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TEMPORAL DYNAMICS OF FOREST PATCH SIZE DISTRIBUTION AND FRAGMENTATION OF HABITAT TYPES IN PENNSYLVANIA

A Dissertation in

Ecology

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Joseph A. Bishop

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The dissertation of Joseph A. Bishop was reviewed and approved* by the following:

Wayne L. Myers
Professor of Forest Biometrics
Dissertation Co-Advisor
Co-Chair of Committee

Robert P. Brooks
Professor of Geography and Ecology
Dissertation Co-Advisor
Co-Chair of Committee

Richard H. Yahner
Professor of Wildlife Conservation

Shelby J. Fleischer
Professor of Entomology

Douglas A. Miller
Associate Professor of Geography

David M. Eissenstat
Professor of Woody Plant Physiology
Chair, Intercollege Graduate Degree Program in Ecology

*Signatures are on file in the Graduate School.

ABSTRACT

Pennsylvania forests have been reported to be changing inefficiently in terms of developed land change per unit population. Change was quantified through two eras of satellite-derived land cover data separated by nine years, 1992 and 2001. Validation of these data revealed accuracies between forest and non-forest classes greater than 80%, 83% and 88%, respectively. Forest change that occurred between the two eras was analyzed to determine our ability to detect forest change and its impacts on avian habitats. Mapping revealed increasing forest fragmentation patterns, in some regions, that are potentially detrimental to avian communities.

Forest fragmentation change was tracked using a variety of geographic information system (GIS) techniques. Changes among patch size and between edge and core forest were calculated. Forest cover was classed by collecting contiguous forest patches into size classes ranging from 1 ha to 25,000 ha for each era. Areas that changed patch class were analyzed. More than 60% of small patches (< 10 ha) converted to non-forest while total number of patches in small classes, nevertheless, increased at the expense of patch classes between 25 ha and 100 ha. In contrast, larger patch classes, > 500 ha, tended to remain in their original class. Core versus edge conversion was similar with 34% of previously edge forest changing to non-forest.

Ecological landtype association (LTAs) boundaries were selected to analyze forest change because their size (mean = 1078 ha) and ecological relevance. Fragmentation change metrics were calculated within LTAs. When mapped, most forest change metrics illustrated clear regional fragmentation trends. Frequently, LTAs clustered together showing regions undergoing similar fragmentation. Notable regions include north-central Pennsylvania, showing evidence of fragmentation decrease, and northeastern Pennsylvania showing patterns of increasing fragmentation.

To understand the impacts of forest fragmentation on natural habitats, breeding bird survey (BBS) data were used to create forest and grassland guilds to test avian responses. Avian guild richness responded predictably to fragmentation change for all guilds and the grassland area sensitive guild had significant results. With the majority of larger, more stable, forest patches under public ownership, consequently, results help to emphasize the management challenges Pennsylvania faces when managing smaller, privately owned, woodlots.

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Chapter 1

History and Basis for the Study of Landscape Change in Pennsylvania

1.1 Introduction

Ecological disturbance research of the shifting-mosaic steady-state has been one line of inquiry in landscape ecology that directly includes time (Forman & Godron 1986; Baker 1989a, 1989b). Disturbance occurs at all ecological levels (e.g., ecosystem, community, population, etc.) and is scale dependent in its detectable influence (White & Pickett 1985; van der Maarel 1993). For example, the influence of a single tree-fall is undetectable at an ecosystem level while, at the community and population levels, it has a detectable influence. In contrast, fire is an important process of grassland ecosystems. A disturbance is any event that disrupts the natural ecosystem, community, or population structure affecting resources and their availability (White & Pickett 1985). Many disturbances appear devastating until the larger landscape is studied. In reality, disturbance is one factor that drives diversity and is an important force in landscape structure (van der Maarel 1993). The *Intermediate Disturbance Hypothesis* states that "the highest level of diversity in a community subject to disturbance is maintained under intermediate levels of disturbance. The lowest diversity is found at both ends of the successional process either immediately following a disturbance or once that community reaches its climax state" (Connell 1978; van der Maarel 1993). In the larger landscape, patch cycling occurs where multiple successional states exist allowing for plant regeneration and animal movement that maintain a regional landscape in a sustained condition. It is the internal patches in the regional landscape that shift through time from one successional state to the next.

Disturbance ecology includes both natural and anthropogenic disturbances and seeks to determine how disturbances affect ecological systems. Natural disturbances tend to be more temporary, while, human-induced disturbances are more permanent. Natural disturbances, such as wind and fire, are influenced and, at times, replaced by human management (de Blois 2001). Naturally occurring environmental constraints are influenced by land use history. In managed landscapes, disturbance patterns are influenced by the best soils and slopes for agriculture and the best locations for urban/suburban development. Human activities often produce sharper edges between forests and developed areas than edges created by natural disturbances (de Blois 2001; Bresee et al. 2004). Then, unlike natural disturbances, anthropogenic landscape changes seldom are allowed to return to their original state. These disturbances are dominated by agricultural and urban and suburban land uses and actively are maintained in the disturbed condition (Bishop & Myers 2005). Trends have shown that these disturbances tend to become more permanent, thus, less likely to return to a natural state (Montaigne 2000; NRCS 2000; Brown 2004).

1.1.2 Temporal Dynamics in Relation to Scale

Temporal dynamics of landscapes have been shown to have important implications for habitat quality (Harris 1984; Forman & Godron 1986; Levins 1992; Wiens 1989; Turner 1989; Turner 1990; Harris 1988; Wiens 2002; ELI 2003). Knowing not just that an area changed, but knowing how it changed and the rate at which it changed helps to understand the likelihood of future change. Including the time, or spatiotemporal (space and time), variable in landscape ecology studies leads to using the discovered trends in management decisions. Time should be accounted for in ecological studies because a single look or measurement is not sufficient to detect the potential for future change. Thus, estimating rates of change or vulnerabilities to change created by an initial change would be impossible (Forman & Godron 1986; Baker 1989b; Sklar & Costanza 1991; Muller & Middleton 1994; Peuquet 1999; Henebry & Merchant 2002) and knowing what time period a recorded event took place and the mechanism to analyze the spatiotemporal results are necessary.

Time is one of three aspects of scale that are important to know when studying landscape patterns. The other two are *grain* and *extent* (Forman & Godron 1986; Turner 1989; Wiens 1989; Turner 1990; Levin 1992; Turner et al. 2001; Wiens 2002; Bennett 2003; ELI 2003). Grain is the resolution at which data are collected. Extent is the total area over which a study takes place. For example, the primary data sources for this study are LandSat TM (Thematic Mapper) and ETM (Enhanced Thematic Mapper) satellite mounted sensors. These sensors collect data for 30 x 30 m square cells on the ground, thus, grain of the sensor is 30 x 30 m (900 m^2). In contrast, a study based on field collected data may calculate plant diversity within a 1 m x 1 m plot so its grain is (1 m^2). For extent, you might be studying all birds nesting in a 10 ha forest patch so your extent would be that 10 ha patch and your grain would be the location of each nest. In contrast, you may be studying the movements of those nesting birds as they forage by moving between their nesting forest patch to other forest patches scattered throughout a valley. In this case, your extent would be the total area used for nesting and foraging.

This study uses a geographic information system (GIS) for spatial data analyses. A GIS is a collection of hardware and software specifically designed to store, update, and analyze georeferenced data (Bernhardsen 1992; Lillesand & Kiefer 1994). There are two formats that are used to represent data within a GIS; vector and raster. Vector data can be visualized and organized by the basic components of points, lines, and polygons represented by x, y coordinates. Raster data are represented by evenly spaced cells where, with the known spacing, x, y coordinates can be derived. One method used to add the third dimension, z-coordinate, to a GIS is with a digital elevation model (DEM), a raster layer with elevations as cell values. Time adds the fourth dimension to these GIS topologies. Peuquet (1999) and Henebry and Merchant (2002) describe three concepts of spatiotemporal data representation of discrete events in GIS: location-based, entity-based, and time-based. A representation of a location-based mode involves sequential data layers with each layer being a "snapshot" in time. The entity-based mode treats geographic objects (e.g., points, lines, and polygons) as variables in time as topological vectors. The time-based mode uses temporal vectors as events that generate a temporal topology. This gives time a direction (Peuquet & Duan 1995; Peuquet 1999; Henebry & Merchant 2002). Throughout my study, a location-based mode is used. Two "snap-shots" in time are compared to assess the landscape level change that occurred between two dates approximately nine years apart. Landscape change that occurred between these two dates is explored to reveal relationships of temporal dynamics to habitat condition (Sklar & Costanza 1991; Henebry & Merchant 2002).

1.1.3 Importance of Land Cover Data for Forest Habitat Detection and Fragmentation Change

The Landsat satellite program was specifically designed to address natural resource management applications. The first satellite in the program was put into orbit in 1972, and six more satellites have been built and launched. Beginning with the fourth satellite, Landsat's primary sensors, the Thematic Mapper (TM) and, launched in 1999, the Enhanced Thematic Mapper (ETM), collect data at a 30 m x 30 m (900 m^2) resolution (Lillesand & Kiefer 1994). TM and ETM data have been used for countless projects to assess habitat condition and landscape analyses (Scott et al. 1993; Lillesand & Kiefer 1994; Loveland & Shaw 1996; Jones et al. 1997; Myers et al. 2000; Bissonette 2003; Bishop & Myers 2005). The nationally structured habitat detection program known as Gap Analysis Program (GAP) relies on these data as a primary data source. Each state project constructs a vegetative land cover layer that is used as a modeling basis to locate potential habitat for each vertebrate species. The models are then used to predict areas possessing high biodiversity to help inform management decisions (Scott et al. 1993; Myers et al. 2000). For this study, forest patch size and core versus edge forest designations have been processed for two land cover classifications of Pennsylvania separated by approximately nine years. Analyses for this study were conducted to detect the changes to available forest habitat and habitat structure that occurred between 1992 and 2001 (Myers et al. 2000; Myers & Warner 2003; O'Connell et al. 2002; O'Connell et al. 2007).

1.1.4 Important Concepts and Terms for this Study

<u>Response guilds:</u> "Response guilds are composed of taxa that respond predictably to disturbances. The members of an individual response guild would respond similarly to a

given disturbance. Response guilds can be generated from species lists, thus, describing what species are present, not just how many are there" (O'Connell et al. 1998).

<u>Biological Integrity vs. Habitat Integrity</u>: Biological integrity has been described as: "The capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region. A system possessing integrity can withstand and recover from most perturbations imposed by natural environmental processes, as well as many major disruptions induced by man." (Karr 1981; Angermeier & Karr 1994). In the context of this study, response guilds were identified to serve as surrogates for biological integrity. Guild richness does not satisfy the essence of biological integrity as described by Karr (1981). However, guild species richness in an area for a particular guild would indicate habitat that meets the needs of many members of that guild (O'Connell et al. 1998; Bishop & Myers 2005). Thus, the term *habitat integrity* will be used to signify areas of quality habitat for a particular guild. An area of high species richness for a response guild will be said to have habitat integrity for that guild.

<u>Habitat Fragmentation:</u> Habitat fragmentation is the breakup and conversion of extensive habitats into smaller isolated habitat fragments too small to support their original species compositions (MacArthur & Wilson 1967; Wilcove 1987; Myer 1994). Harris (1984) notes two components of fragmentation as: (1) conversion of natural habitat in a landscape to other covers; and (2) separation and isolation of the remaining natural habitat into smaller patches.

<u>Landscape Connectivity:</u> Structural vs. Functional – Landscape Connectivity has been defined as "the functional relationship among habitat patches, owing to spatial contagion of habitat and the movement responses of organisms to landscape structure" (With et al. 1997). Connectivity is species dependent and is not simply based on physical habitat connections alone. In this study, I address the physical connections or *Structural*

Connectivity which is the physical adjacency of forest patches, when I class forests into forest patch sizes. Then, I address *Functional Connectivity* of landscapes where forest patches are connected by the movement (foraging and dispersal) abilities of species that use the habitats.

1.1.5 Study Area: The Landscapes of Pennsylvania

Pre-European Pennsylvania was estimated to have been between 90% and 95% forested (Rhoads & Black 2005). Grassland patches of 100 ha or more were scattered throughout the state and resembled the prairies of the Midwest (Schein & Miller 1995; Rhoads & Black 2005). Forest clearing reached its maximum between 1890 and 1930. The once forest dominated state was reduced to approximately 32% forest cover (Rhoads & Black 2005) and some report as low as 10% (S. Hoffman, personal communication 2004). Exact totals vary, but of the original forest only a few thousand hectares of *Ancient Forest* remain and many areas have been logged more than once. Much of the deforested land was not suitable for agriculture and was left to regenerate. Today approximately 60 to 65% of the state is forested, but this second- and third-growth forest is not the same as the original with regard to structure or composition. Pennsylvania's history of timber production with 50 m tall white pines (*Pinus strobus*) and eastern hemlocks (*Tsuga canadensis*) with a mature understory has been replaced by even-aged forest stands less than 80 years old (Schein & Miller 1995; 21st Century Environment Commission 1998; Rhoads & Black 2005). In addition to the logging impacts on the forests of Pennsylvania, several pests have influenced them as well. In 1911, chestnut blight was first reported and would ultimately lead to the elimination of nearly all native American chestnut (*Castanea dentata*) trees. Also, pests such as gypsy moth (*Lymantria dispar*) and, more recently, hemlock woolly adelgid (*Adelges tsugae)* are affecting forest health (Rhoads & Black 2005).

Gradually, over the past 100 years, the landscapes of Pennsylvania have become human dominated, generally, following one of two paths of change. Lands developed either by first having been cleared from forest, put into agricultural land and then being converted into urban/suburban land or, secondly, by skipping the agricultural phase and proceeding directly from forest to urban/suburban areas. In contrast, much of the previously deforested north-central Allegheny Plateau, high elevations of the Pittsburgh Low Plateau, locally known as the Laurel Highlands, and the ridge-tops in the Ridge and Valley have been allowed to reforest primarily due to the difficulty of farming on steep slopes and the productively poor agricultural soils (Fig. 1.1)(Schein & Miller 1995; Rhoads & Black 2005). Reforestation spanned from the late 1800s, when only about 10% of Pennsylvania was forested, to the present, with about 65% forest cover (21st Century Environment Commission 1998; Goodrich et al. 2002). During the same time period the regional development trend has been to convert previously farmed land into urban/suburban lands (Vitousek et al. 1997; Montaigne 2000; NRCS 2000). NRCS (2000) reported that for all of the Piedmont ecoregion and large portions of the Glaciated Pocono Plateau, Glaciated Low Plateau, and Pittsburgh Low Plateau of Pennsylvania 49.6 % of the area that was forest in 1982 converted to non-forest by 1997.

Forests of Pennsylvania have regenerated and developed into habitat for a wide variety of animal species. Species that were reduced or eliminated, as the forests were cleared, returned as forests regenerated. More than 71 (38 forest only) species of birds, 43 species of mammals, and 48 species of reptiles and amphibians rely on forests in Pennsylvania for essential habitat. Many of these species rely on large, relatively contiguous, forested areas for their continued survival (Myers et al. 2000; Goodrich et al. 2002; Moyer 2003). Partners-in-Flight (PIF) considers the forests of Pennsylvania critical habitat for 20 species of neotropical migrant birds (Rich et al. 2004). Twenty-eight percent (20/71) of these birds are already listed as threatened, endangered, or in decline. Fifteen (23%) mammals and 16 (22%) reptiles and amphibians are listed as well (Peterjohn et al. 1995; Hulce 1998; Wright & Kirkland 1998; S. Hoffman, personal communication 2004; A. Linzey, personal communication 2004; Rich et al. 2004; C. Mahan, personal communication 2005).

Uncontrolled logging of the 19th century was replaced by urban and suburban growth in the 20th century as the most critical threat to wildlife habitat. Berks, Montgomery, and Chester counties in southeastern Pennsylvania and Pike and Monroe counties in northeastern Pennsylvania, based on population increase, are among the fastest growing counties in the United States and Pennsylvania has been reported to have one of the highest road densities of any state in the country (Montaigne 2000). Several recent reports have been critical of the development in Pennsylvania, calling it "inefficient". The Governor's 21st Century Environment Commission (1998) reported that between 1960 and 1990, whereas population in the 10 most populated counties grew 13%, the amount of land used grew 80%. Also, an independent document prepared by the Brookings Institute (2003) reported that from 1982 to 1997 the population of Pennsylvania grew 2.5%, but its urbanized footprint grew 47%. This means that the third slowest growing state developed the sixth largest amount of land. The report stated that "Pennsylvania is squandering a key source of competitive advantage: its natural assets".

1.2 Chapter Overviews

Chapter 2 - **Habitat Patch Change Analysis over Two Temporal Frames**

- *Did forest fragmentation patterns change between 1992 and 2001?*
- *How did forest patch sizes and their distributions change across Pennsylvania?*

Central to the disciplines of conservation biology and landscape ecology are the study, understanding, and identification of landscape characteristics, such as; habitat connectivity, corridors, and contiguous habitats (Harris 1984; Noss 1987a; Simberloff & Cox 1987; With et al. 1997; Taylor et al. 1993; Bennett 2003), habitat islands and dispersal distances (MacArther & Wilson 1967; O'Connell et al. 1998; O'Connell 2000; Bennet 2003; ELI 2003; Keller & Yahner 2007), habitat cohesion (Turner 1989; Levins 1992; Opdam et al. 2003), and core habitat vs. edge habitat and associated edge effects (Brittingham 1983; Harris 1988; Yahner 1988; Debinski & Holt 2000). These characteristics, when coupled with their potential impacts, have been applied in many management scenarios. Directly tied to their study and understanding is scale. Each of the above listed landscape characteristics are scale dependent, meaning that measurements will vary depending on the grain and extent at which the measurement was observed (Turner 1989; Wiens 1989; Levins 1992; Wiens 2002). This is important because in any ecological study it is necessary to ask: Is the scale that measurements are being recorded appropriate to address the question being asked?

Goodrich et al. (2002), based on 1992 land cover data, assessed the landscape of Pennsylvania focusing on the condition of wildlife habitat. Core and edge forests were identified and connected forest blocks were grouped into six area categories related to the varying habitat needs of wildlife. They reported that 57% of the forested area of Pennsylvania was actually edge forest within 100 m of a disturbance or anthropogenically maintained land cover. The present study built on that assessment by, first, duplicating those calculations using new land cover data from 2001 and then, secondly, analyzing forest change to better understand forest condition. These data were then classified, for both dates, based on area thresholds that relate to wildlife habitat needs (Jones et al. 1997; ELI 2003; O'Connell et al. 2008). Graphics and tables were constructed to help identify trends in forest patch change that occurred between 1992 and 2001.

Three data layers were created for both, 1992 and 2001. These layers are core vs. edge forest, forest size patch class, and forest connectivity. Each layer characterizes forest areas according to forest condition, patch size, and connectedness. The core vs. edge forest data layer separates core from edge forest. Edge forest is the forested strips, or buffers, that are within the first 100 m or three 30 x 30 m raster cells into forest from the edge of a disturbed or anthropogenic land cover (Robbins et al. 1989; Debinski & Holt 2000; Goodrich et al. 2002). Processing for forest patch size class simply groups forest into one of 15 forest classes based on area of connected forest. Classes range from the smallest patch area of 1 ha, to 25,000 ha. Forest patch size classes increase based on approximate doubling of area 1 ha, 2 ha, 5 ha, 10 ha, 25 ha, 50 ha, etc. Size classes follow habitat thresholds described by Environmental Law Institute (2003). Two types of forest connectivity were addressed. Structural connectivity as forest that is physically connected to other forest patches or, functional connectivity as forest that is within a predetermined distance away from other forest patches. Functional connectivity was calculated using a *focal function* in the Spatial Analyst extension to ArcGIS software

(ESRI 2006) that tabulated percent forested areas from within predetermined distances from every point location in Pennsylvania. Distances were selected based on predicted mobility of groups of wildlife species and the likelihood that they could move between forest patches within movement thresholds (ELI 2003). Each of these three layers then were differenced and mapped.

Validation of these data revealed accuracies between forest and non-forest classes greater than 80%, 83% for 1992 and 88% for 2001. Results of the validation were best when focusing on the forest versus non-forest designation. When attempting to more precisely interpret land cover our classifications were good but a little less reliable.

Mapping revealed forest change patterns that appear to be increasing in some regions that are potentially detrimental to avian communities. Forest fragmentation change was tracked using a variety of geographic information system (GIS) techniques. Changes among patch size and between edge and core forest were calculated. More than 60% of small patches (less than 10 ha) converted to non-forest while total number of patches in small classes, nevertheless, increased at the expense of patch classes 25 ha, 50 ha, and 100 ha. In contrast, larger patch classes, greater than 500 ha, tended to remain in their original class. Core versus edge conversion was similar with 34% of previously edge forest changing to non-forest, while 71% of the original (1992) core forest remained core forest in 2001.

Chapter 3 - **Landscape Characteristics of Forest Fragmentation Change as Captured by Ecological Landtype Associations**

- *Do landscape characteristics reveal fragmentation patterns?*
- *Do forest change intensities vary across Pennsylvania?*
- *Can areas of similar change intensity be logically grouped?*

Bailey (1983) and Omernik (1987), then further refined by Bailey (1995); and Omernik (1995) described and delineated ecoregions of the United States. These were defined at continental and regional scales and incorporated climatic regimes and physiographic features. The U.S. Environmental Protection Agency adopted Omernik (1995) and participates in its further refinement as part of the North American Commission for Environmental Cooperation (CEC) formed in 1997 (National Atlas 2007). The U.S. Forest Service adopted the National Hierarchical Framework of Ecological Units (ECOMAP 1993) classifying eight levels of ecological units starting with Bailey's ecoregions, at the broadest scale, and then sequentially nesting each level into finer scaled units. The procedures followed by Bailey and Omernik for ecological mapping are similar and combine physical features, such as climate, topography, physiography, and geology with organic features, such as soils, vegetation, and wildlife to help delineate boundaries. The Forest Service, in cooperation with the Pennsylvania Bureau of Forestry, delineated the first five levels of units for Pennsylvania (Myers 2000; Kong 2006).

Kong (2006) delineated ecological unit level six, the Landtype Association (LTA). As defined by Myers (2000), *"LTAs are complexes of complementary landscape features that combine through spatial adjacency to create ecological contrasts across regions. These factors affect biotic distributions, hydrologic function, natural disturbances and land use patterns."* Kong (2006) based the LTA delineations on topographical features that incorporated both hydrologic and habitat conditions with the assumption that these features capture ecological characteristics at a landscape scale. This delineation divided the state into two physically structuring formations termed *Caplands* and *Cuplands*. *Caplands* capture physical structures that arch upward and *Cuplands* capture the sagging to planar physical structures. These subdivide into 18 additional classes; nine *Caplands* and nine *Cuplands,* incorporating local and regional features within their delineations. Kong (2006) validated the delineations with data from stream networks, land cover patterns, and vertebrate habitats and the ecological units captured much of the variation in these data. The final LTA data layer contains more than 10,000 polygons grouped into one of the 18 classes. This study uses the LTA delineation to help identify and characterize areas of varying forest change intensity.

Assessment of the forest patch data (Chapter 2) revealed four visually identifiable patterns of fragmentation change. Fragmentation patterns and example locations include: (1) intense urban/suburban sprawl, both the Philadelphia and Pittsburgh suburban regions display obvious changes, (2) an increase of forest patch class size in active forest management areas in northern Pennsylvania immediately east of the Allegheny National Forest, (3) new agricultural expansion in the glaciated region of northern Pennsylvania north-northwest of the city of Scranton, and (4) suburban sprawl in the Ridge and Valley ecoregion, where valleys contain many areas for which the separation and size of forest patches appears to be changing.

Such patterns help to identify and characterize forest areas experiencing varying rates of change. Forest size class composition was tabulated for each data layer (1992 and 2001) within each LTA. These values then were differenced to capture change in forest patch size composition. Forest change distribution was evaluated to locate areas of similar forest change intensity. LTA polygons exhibiting similar change intensity were grouped together for further analysis.

Areas displaying similar forest change intensity were described by a suite of landscape metrics. Landscape metrics were selected that have proven to be informative when studying habitat fragmentation, based on past studies (Debinski & Holt 2000; Riitters et al 2000; Jones et al. 2001; Goodrich et al. 2002; Li & Wu 2004; R. Gardner, personal communication 2006). In contrast to many studies exploring landscape metrics, R. Gardner (personal communication 2006) stressed a strategic selection of metrics that would be appropriate to goals of this study. The landscape metrics selected for this study, are listed here along with the fragmentation characteristic that each is reported to inform: (1) change in edge, measured by change in area weighted edge density (AWED), (2) changes in forest area, measured by the change in mean forest patch size (MPS), (3) changes in the forest edge to forest patch area ratio, as captured by a change in fractal dimension (FD), (4) change in the distance between forest patches, as captured by the change in the median distance between patches (DIST), and (5) change to core forest habitat, as captured by a changes in the core vs. edge habitat composition (McGarigal & Marks 1995; Goodrich et al. 2002).

Ecological landtype associations (LTAs) boundaries were selected to analyze forest change because their size (mean = 1078 ha) and ecological relevance. When mapped, most forest change metrics illustrated clear regional fragmentation trends. Frequently, LTAs clustered together showing regions undergoing similar fragmentation trends. Notable regions include north-central Pennsylvania, showing evidence of fragmentation decrease, and northeastern Pennsylvania showing patterns of increasing fragmentation. Areas of Pennsylvania, that we predicted to be undergoing increased fragmentation, such as the areas surrounding Philadelphia and Pittsburgh, also displayed regional patterns as captured by the LTAs.

Chapter 4 – **Breeding Bird Response to Fragmentation Change in Pennsylvania**

- *What does forest patch size change reveal about habitat integrity based on Breeding Bird Survey data change?*
- *Does change in guild species richness correlate with forest patch size change?*
- *Do changes in Breeding Bird Survey data suggest habitat thresholds for the bird guilds?*

Grouping of species into guilds that share similar behavioral and/or habitat needs has been proposed in past studies. Functional, compositional, trophic, structural, and response guilds have all been suggested and tested in habitat inventory and management scenarios (Severinghaus 1981; DeGraaf et al. 1985; Croonquist & Brooks 1991; O’Connell et al. 1998; O’Connell et al. 2000; Goodrich et al. 2002; Bishop & Myers 2005). Response guilds group species by their predicted response to a habitat change (O’Connell et al. 1998). For this study, four response guilds were proposed to assess habitat integrity for Pennsylvania. The guilds tested include grassland area sensitive, grassland habitat, forest interior habitat, and large forest obligate (O’Connell et al. 1998; Goodrich et al. 2002; Bishop & Myers 2005). Scale of resource use and movement and dispersal behaviors can be accounted for in response guilds. In this case, the response being tested is associated with changes in habitat area and habitat fragmentation. The landscape metrics collected in Chapter 3 are well suited for testing habitat integrity as relating to habitat size and connectivity. Habitat integrity for a guild was defined by species presence for at least 50% of the guild members (Bishop & Myers 2005; O'Connell et al. 2007). To validate these analyses, areas of varying habitat integrity, based on changing fragmentation, were compared to breeding bird survey (BBS) trends recorded over the past nine years.

Breeding Bird Survey (BBS) data were compared for each 3-year period that coincides with the two eras of the satellite imagery. BBS routes were chosen from within areas of similar forest change intensity as identified in Chapter 3. BBS data are distributed in 10-stop segments along each BBS route. Thus, five 10-stop summaries are available for each 50-stop survey route. A stratified sample of 10-stop BBS segments was selected from areas that displayed similar forest change intensity. As compiled by O'Connell et al. (2007), landscape variables were collected from within a 500-m buffer zone around each 10-stop segment and then analyzed along with the same variables from the forest change intensity areas.

To understand the impacts of forest fragmentation on natural habitats, breeding bird survey (BBS) data were used to create forest and grassland guilds to test avian responses. Avian guild richness responded predictably to fragmentation change for all guilds and the grassland area sensitive guild had significant results.

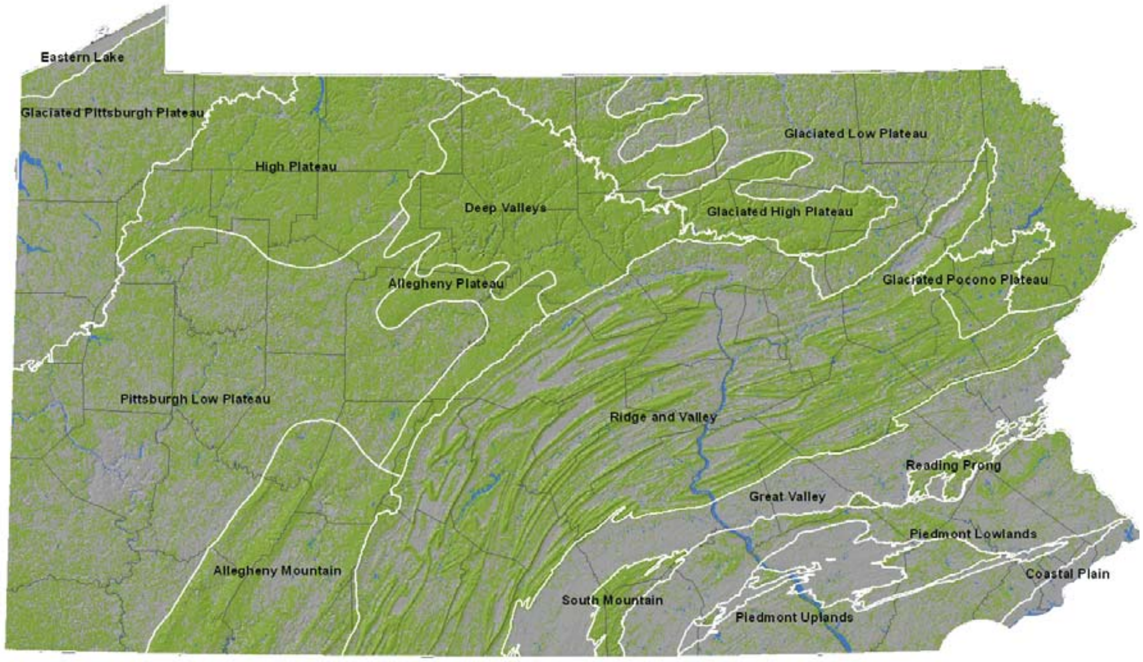


Figure 1.1 – Forested areas from 2001, in green, with shaded elevation data to help display landform. White boundaries delineate Level-3 Ecological Regions (Bailey 1995).

Chapter 2

Habitat Patch Change Analysis for Two Temporal Frames

2.1 Introduction

2.1.1 Habitat Fragmentation

Habitat fragmentation has been described as the breakup and conversion of extensive habitats into smaller isolated habitat fragments too small to support their original species compositions (MacArthur & Wilson 1967; Wilcove 1987; Myer 1994). Harris (1984) notes two components of fragmentation as: (1) conversion of natural habitat in a landscape to other covers; and (2) separation and isolation of the remaining natural habitat into smaller patches. Habitats of Pennsylvania have been reported to be changing inefficiently with respect to ratios of developed land area to population (21st Century Report 1998; Goodrich et al. 2002; Brookings 2003; Moyer 2003). Forest habitat fragmentation has been identified as one negative symptom of this inefficient change and many areas appear to be fragmenting more rapidly. For example, Brookings (2003) reported that from 1982 to 1997 Pennsylvania population grew 2.5% but its urbanized footprint grew 47%, meaning that the third slowest growing state developed the sixth largest amount of land. For this study, I only address fragmentation of forested habitats, but other habitat types, such as grasslands, wetlands etc., could be studied in the same manner.

As fragmentation progresses, maintaining connectivity of habitats becomes critical to the sustainability of wildlife populations found within a landscape (Noss 1987a; Noss 1987b; Simberloff & Cox 1987; Saunders et al. 1991; Noss & Csuti 1994; Bennett 2003). Metapopulation theory suggests that a species ability to move among habitat patches and utilize appropriate habitats is critical to that species survival (Hansson 1991; Opdam 1991; Hanski & Simberloff 1997; Hanski 1999). A metapopulation has been defined as a system of populations of a species in a landscape linked by balanced rates of extinction and colonization (Pickett & Rogers 1997). Disturbances can alter this balance by affecting a species ability to move in a landscape. Natural disturbances are temporary, often ecologically necessary, impacts that can cause shifts within an ecological system (e.g., fire, wind). Following a natural disturbance, under normal conditions, animal species shift their habitat use to adjacent areas during regeneration (Garton 2002). However, anthropogenic disturbances often are permanently maintained conditions interfering with natural regeneration and previously resident species are prevented from re-colonization, thus permanently altering species composition (Pickett & Rogers 1997; Hanski 1999). As anthropogenic disturbance occurs and expands it becomes more difficult for the original resident population of a species to find appropriate habitat. Studies have shown that as fragmentation increases, it will eventually isolate habitats making it difficult for wildlife to forage and disperse among the remaining habitat patches (MacArthur & Wilson 1967; Harris 1984; Harris 1998; Saunders et al. 1991; Vos et al. 2002; Bennet 2003; ELI 2003; Keller & Yahner 2007).

Many avian species, for example, show specific responses to shrinking habitat patches. Once patch size is reduced beyond a species minimum area threshold, nesting and foraging success are affected and eventually some species no longer use the reduced habitat patches that remain (Temple 1984; Blake & Karr 1987; Robbins et al. 1989; Bolger et al. 1997; O'Connell et al. 1998; O'Connell et al. 2008). Along with general habitat shrinkage, edge habitat expands and brings increasing habitat pressures brought on by generalist and parasitic species (Brittingham & Temple 1983; Yahner 1988; Rodewald & Yahner 2001; Yahner 2003).

Maintaining or creating corridors to preserve connections between larger habitat patches has been shown to benefit wildlife populations (Harris 1984; Wilcove et al. 1986; Noss 1987a; Saunders et al. 1991; Vos et al. 2002; Bennet 2003). With et al. (1997) define landscape connectivity as "the functional relationship among habitat patches, owing to spatial contagion of habitat and the movement responses of organisms to landscape structure". This suggests that connectivity is not solely based on landscape structure (Taylor et al. 1993), but also relates to the ability of a species to move among habitat patches that may be disconnected, but within the threshold distance for an organism to move, thus functionally connected (With & Crist 1995; Gustafson & Gardner 1996; Pearson et al. 1996; With et al. 1997). This establishes the difference between structural connectivity, where habitat is physically connected, and functional connectivity, where species ability to use habitat is not interrupted by short separations between habitat patches. These connections maintain the contiguity of habitats preventing habitat islands from forming and facilitating dispersal (Bennet 2003; ELI 2003; Keller & Yahner 2007). Opham et al. (2003) described a method of assessing habitat cohesion that accounted for the combination of true habitat connection as structural connectivity, with short habitat separations easily traversed by mobile species as functional connectivity. This habitat cohesion represents the connectedness of habitat for a particular species at a particular scale.

Vital to understanding the limitations of a study is accounting for the spatial scales that may be influencing the study. Each of the landscape characteristics mentioned above connectivity, habitat edge, cohesion, etc., is scale dependent (Turner 1989; Wiens 1989; Turner & Gardner 1990; Levins 1992; Wiens 2002), meaning that calculation of a landscape metric and reliability of detecting a change in that metric is dependent on the extent of area over which the metric is measured and the grain of the original data. Metric values vary depending on the scale that measurements were taken (Forman & Godron 1986; Levins 1992; McGarigal & Marks 1995; Riiters et al. 1995; Riitters et al. 1997; Vos et al. 2001; Opdam et al. 2003; Riitters & Wickham 2003; Wade et al. 2003; Liu & Wu 2004; Neel et al. 2004). This is important because in ecological studies the scale that measurements are being recorded needs to be appropriate to address the goals of the study. In this study, spatial scale is directly connected to the base land cover data. All forest interpretations are based on these data.

Research Objectives

- *Can forest change be detected by remote sensing satellites based on the two land cover interpretations for Pennsylvania?*
- *How has fragmentation changed in Pennsylvania during the nine years of separation between the two dates of the land cover?*
- *Can forest change be characterized? (Between forest and non-forest, and between forest patch size classes)*
- *What are the size-specific patterns of forest change?*

2.1.2 Land Cover Classification

The Landsat program was initiated in 1967 and put its first satellite into orbit in 1972. To date, six Landsat satellites successfully have been launched and planning for a new satellite is underway (Lillesand & Kiefer 1994; USGS-LCI 2007). The Landsat program was designed to address natural resource and land management applications, and the onboard sensors were selected and configured specifically to meet these needs. Landsat 4 was the first to carry the Thematic Mapper (TM) sensor. The TM sensor improved on the original Multi-Spectral Scanner (MSS) by collecting data in seven bands or wavelength ranges of the spectrum and improving the pixel resolution to 30 x 30 m (900 m^2). Total scene area is 185 x 185 km, and these sensors were designed and have proven to be useful particularly for distinguishing among vegetation cover types (Lillesand & Kiefer 1994). Landsat satellites 5 and 7 are still in operation, Landsat 6 failed at launch (Lillesand & Kiefer 1994).

The Multi-Resolution Land Characterization (MRLC) consortium of federal agencies was formed to establish a consistent land cover data set for the United States while preserving financial resources (Loveland & Shaw 1996; Myers et al. 2000). The goal of the MRLC is to provide data across a range of spatial and temporal scales for analysis and monitoring of environmental change (Loveland & Shaw 1996). The 1992 National Land Cover Dataset (NLCD) was the first product released by the MRLC and the NLCD just released the 2001 land cover data in January, 2007 (Homer et al. 2004).

Goodrich et al. (2002) used 1992 land cover data to assess the landscape condition of Pennsylvania emphasizing threats to wildlife habitat. Core and edge forests were identified and forest patches were grouped into six size categories that related to the varying habitat needs of wildlife. Edge forest was defined as forested land within the first 100 m from a non-forest land cover (Robbins et al. 1989; Debinski & Holt 2000; Goodrich et al. 2002). They reported that 57% of the forested area in Pennsylvania was actually edge forest within this 100-m zone.

This study builds on that assessment by reevaluating their work and applying their procedures and calculations to new land cover data collected in 2001. These data were then grouped into forest patch sizes classes for both dates, and core versus edge forest then related to wildlife habitat needs (Jones et al. 1997; O'Connell et al. 2000; ELI 2003). Forest patch size changes were compared to identify the habitat changes that occurred between the two dates.

2.2 Methods

2.2.1 Land Cover Validation

Two land cover classifications were compared for this study. The earliest data were created by the Pennsylvania Gap Analysis (PAGAP) project (Myers & Bishop 1999; Myers et al. 2000). This classification was created using data from the Thematic Mapper (TM) sensor on Landsat 5 that has a 30-m pixel resolution. Data from 10 satellite scenes are required to cover Pennsylvania. The scenes selected were recorded between 1991 and 1994 (later referred to as the 1992 land cover data) (Fig 2.1). These scenes were originally chosen by the Multi-Resolution Land Characteristics (MRLC) consortium (Loveland & Shaw 1996; Scott et al. 1993). The initial Pennsylvania classification, created during PAGAP, included eight land cover classes and then incorporated a separate three class urban versus suburban interpretation. These two layers were cross-classified, within a geographic information system (GIS), to combine the two classifications creating the final 24 category land cover layer (Table 2.1). For the more recent land cover data, satellite scenes were recorded between 1999 and 2002 (later referred to as the 2001 land cover data) by the Landsat 7 Enhanced Thematic Mapper (ETM) sensor. These data have the same 30-m pixel resolution, as the 1992 data, and were classified into 15 land cover categories (Table 2.1)(Myers & Warner 2003).

Goodrich et al. (2002), the Brookings Institute (2003), and Partners-in-Flight (Rich et al. 2004) have commented on how important forested habitats are to Pennsylvania. For that reason, I have focused this study on forest cover classes for the two time frames while combining all other land cover classes into one non-forest category (Table 2.1). This creates a two category GIS data layer for each era, 1992 and 2001, to focus analysis on forest change. One group which is dominated by the three forest classes, but also includes wetland classes will be referred to as *naturalistic,* and the other category encompassing all human influenced land covers (i.e., agriculture, pasture, and urban and suburban development) will be referred to as *humanistic* (Bishop & Myers 2005).

During the PAGAP project a validation of the vegetation classification was conducted to test for the accuracy of the eight land cover classes (Myers et al. 2000). For this study, the results from the PAGAP validation were collapsed to compare just the forest versus non-forest distinction (USGS 2000; Stehman et al. 2003). For the PAGAP validation air videography data were collected along evenly spaced north-south flight lines across Pennsylvania. Randomly placed point locations were then sampled along these flight lines to validate the land cover. At each random location, video was interpreted using the same land cover categories and then compared to the classified satellite data to determine accuracy of the land cover classification (Fig 2.2).

The same flight lines and random points used for the 1992 land cover validation were used to validate the 2001 land cover. However, digital aerial photography (Digital Ortho Quarter Quads or DOQQs) that were collected in 1999 for the eastern half of Pennsylvania were used as ground-truth data (DOQQs can be obtained from the PASDA website at: http://www.pasda.psu.edu). Classification accuracy was assessed in the same manner as the PAGAP 1992 land cover data.

Four versions of validation were conducted for each era to assess accuracies while accounting for various sources of error. Sources of error can be either human interpreter based or machine/satellite based (USGS 2000; Stehman et al. 2003). Interpreter error can be accounted for by pooling classes. This type of assessment groups similar land cover classes together (USGS 2000; Stehman et al. 2003). For example, forest classes (coniferous, deciduous, and mixed) were treated as one cover class if any forest class was detected. Also, in agricultural areas, row crop and pasture were combined and in developed areas both high intensity development and low intensity development were treated as one class. Because this study centers on naturalistic versus humanistic lands, the more subtle differences between all cover classes were of lesser importance.

Two validation tests were conducted to help account for satellite and geographic position errors. These two tests were accomplished by extending the search window to include all raster cells located within a 30-m radius around the sample point. This helps to account for two potential errors. The first error, *conservative bias*, (Verblya & Hammond 1995) arises due to the uncertainty of direct vertical alignment between the airplane mounted GPS unit and the surface of the earth. The second source of error, *optimistic bias*, (Hammond & Verblya 1996) is found in inconsistencies between the spectral values of an individual raster cell and the surrounding landscape matrix where it is located. Within a land cover classification it is common to find individual cells classed differently than the surrounding matrix where the matrix frequently represents the true land cover of those cells. Contingency tables of all validation tests were prepared to facilitate comparison of the results.

2.2.2 Preliminary View – 1992 versus 2001 Comparisons

To test the ability to detect change in forests over time, basic maps were prepared between the two eras. Using ArcGIS (ESRI 2006) software, percent forest was calculated for each cell within a regularly spaced grid spanning Pennsylvania. In this case, the Pennsylvania Breeding Bird Atlas (BBA) (Brauning 1992) project sample grid was selected because it provided a consistent sample area (23.9-24.7 km^2) and because data summarized in these cells would be of use to the BBA project. Percent forest was calculated within each cell from each land cover layer (1992 and 2001). Percent forest area values were subtracted to obtain percent forest change within each cell.

Forest patch area (ha) was processed as a continuous area data layer for each land cover layer. SAS-JMP (2006)(Version 5.1) was used to create cumulative distribution functions (CDF) graphs for each land cover era to show graphically the distribution of patch sizes found across Pennsylvania. For this study, CDF curves show accumulation of state-wide forest area as patch area increases. This predicts the likelihood that a selected forest patch will be a specific patch area. By comparing the two graphs, changes in forest patch distribution between land cover eras can be detected (Hayslip et al. 2004; Hayslip et al. 2006).

2.2.3 Forest Patches Defined

As previously mentioned, this study only addresses the forest classified from two land cover classifications separated by approximately nine years, 1992 and 2001, and treats all other classes as non-forest. Using ArcView 3.3 geographic information system (GIS) software and the Spatial Analyst Extension for ArcView 3.3 (ESRI 2002), the two land covers were first reclassified into binary layers indicating forest and non-forest. Once forest layers were prepared, layers were intersected with road layers to account for roads not consistently detected by satellites alone. Roads layers were selected that best matched the time frame of the land cover data. All paved roads, as well as heavy use dirt and gravel roads, were considered to bisect forest habitat. Low use roads, mostly gated state and federal forest roads, were considered to have little impact on forest habitat and thus, were removed from the roads layers prior to analysis. Roads data that were selected are from 1995 and 2001 and are maintained and available from the Pennsylvania Department of Transportation (PennDOT 1995; PennDOT 2001). Each set of data layers, forest versus non-forest, and roads were combined into one data layer that re-coded forest to non-forest wherever roads passed through a forested area.

Forested areas for each date were grouped into forest patches that connected along neighboring pixels that shared a solid edge. If pixels only touched at corners, they were considered to be in separate patches (Fig. 2.3). Once each layer was separated into patches, patch area was calculated for each forest patch layer and then grouped into forest patch size classes that range from one ha to 25,000 ha. Forest patch classes were established and grouped, within the GIS, according to a doubling in patch sizes. Forest patch size classes are as follows: 1 ha, 2 ha, 5 ha, 10 ha, 25 ha, 50 ha, 100 ha, 250 ha, 500 ha, 1,000 ha, 2,500 ha, 5,000 ha, 10,000 ha, and 25,000 ha. This separation into patch size classes facilitates the detection of patch changes between two imagery eras and helps when evaluating effects on wildlife habitat by using patch sizes as surrogates for home range size estimates and minimum habitat needs reported for many species (Merritt 1987; Robbins et al. 1989; O'Connell et al. 1998; Fergus 2000; Myers et al. 2000; ELI 2003).

2.2.4 Edge versus Core Forest Distinctions

Goodrich et al. (2002) noted that 57% of the forest area of Pennsylvania was classified as edge forest. Increasing edge forest is one symptom of forest fragmentation. Here, I define edge forest as forested area found within the first 100 m into the forest from any disturbance (i.e., agricultural row crop or pasture, suburban or urban development, and roads)(Fig. 2.4)(Robbins et al. 1989; Debinski & Holt 2000).

Using ArcGIS (ESRI 2006) both land cover eras were processed identically to create data layers that isolated core and edge forest from non-forested areas. Creation of this layer took several additional GIS steps beyond the simple reclassification of the land covers required to separate the forest vs. non forest classes. Water takes special handling so that larger water bodies do not artificially connect isolated forest areas or increase edge along a water and forest interface. First, small ponds and wetlands, < 1 ha, and the speckling of water pixels often visible on satellite classifications were treated as their background matrix. Thus, in forested areas, single pixels of water were dissolved into the natural background where in agricultural and developed lands they were dissolved into the non-forest matrix. Next, water speckling found along first through fourth order streams (Strahler 1957) were converted into the background matrix as well. For larger water areas, (e.g., lakes and rivers) water was temporarily removed during the calculation of the 100 m edge distance and then it was added back into the final layer to insure that true water space was represented. Once the initial three category land cover layers were created, the roads layers were used to inscribe the road footprint into the land cover. Roads layers available from Pennsylvania Department of Transportation (PennDOT) were added to account for roads not visible on satellite imagery. Each layer coincided as closely as possible with the date of the satellite imagery (PennDOT 1995; PennDOT 2001). Once this step was complete, the final distance calculations were processed to identify the 100-m edge zones that extend into the forest. Once the two data layers were complete, they were differenced to characterize the change in core versus edge forests.

2.2.5 Forest Patch Change Characterization

To analyze forest patch change, the two forest patch layers were combined into one GIS layer. Each layer was coded so that the patch class values from each date could be distinguished as those from 1992 and those from 2001. Once prepared, these layers were combined to create a summary layer coded in such a way as to reveal the former and current forest patch size class for any location in Pennsylvania. The resulting data layer was analyzed to detect changes in patch size class identifying areas of loss or gain in forest area. This effort was designed not only to detect change, but also to track forest change into its new patch size class. Patch change maps and graphs were created to show distribution of new patch size classes. To help detect trends in patch size change, a contingency table was constructed that separates changes between all patch change possibilities.

2.3 Results

2.3.1 Validation of Land Cover Data

For consistency, I used the same methods to validate the 2001 land cover as were used by Myers et al. (2000) to validate the 1992 land cover data. Although both land cover layers have multiple classes, data were reclassified to isolate and analyze forest versus non-forest. In both cases *naturalistic* areas (forested and wetland classes) were correctly classified at more than 90% of the test points. In 1992, they were 91% accurate, and, in 2001, they were 92% accurate. When including the *humanistic* grouping (agriculture classes and developed suburban and urban classes) and water class, accuracies decreased, but were still within the acceptable limits, near 85%, suggested by Anderson et al. (1976), and Congalton and Green (1999). The 1992 and 2001 classifications were 83% and 88% correct, respectively (Table 2.2)(detailed class validation summaries are in Appendix B).

2.3.2 Results of the Preliminary Look

The first analysis provided a preliminary assessment of where forested areas appear to be changing. Change was apparent as mapped within the Pennsylvania Breeding Bird Atlas grid (Fig. 2.5). Values ranged from a 55% increase in percent forest area to a 58% percent decrease in percent forest area. When examining visible patterns, trends are explainable based on known regional activities. Forest reduction in southeastern and southwestern Pennsylvania reflects the suburban sprawl of Philadelphia and Pittsburgh, whereas the percent forest reduction in the northeast is more likely due to expansion of agricultural areas. What is interesting to note is the increase in forest percent that occurred in the primarily forested north-central portions of the state. This increase in percent forest is likely due to changes in forest harvesting and maturing forest succession, as well as the maturing cover on reclaimed strip-mines.

Using SAS-JMP (Version 5.1), Cumulative Distribution Functions (CDF) were processed for each era (SAS-JMP 2006). The CDF process within SAS-JMP produces two products, the CDF graph as well as a general distribution graph which in this case depicts state-wide distribution of forest patch sizes for each era (1992 and 2001). This analysis was based on a continuous forest area GIS data layer. For both eras graphs depict that the smaller forest patch areas (0-5 ha) comprise a large percentage of the total forest area of Pennsylvania. For 1992, forest patch areas between 0-5 ha comprised 3.8% of the forest in Pennsylvania and, in 2001, they increased to 5.4%. One difference to note is that in 1992 the largest contiguous forest patch is 26,901 ha; whereas, in 2001, the largest forest patch was 17,727 ha. The largest 1992 forest patch was bisected by roads during the 9-year time span. In both cases, the CDF graphs show that between 75% and 80% of the forested areas of Pennsylvania occur in patches less than 2,000 ha (Figs. 2.6 and 2.7).

2.3.3 Forest Patch Class Distribution for Two Eras

Based on these two land cover classifications of satellite data, Pennsylvania had a decrease of 516,263 ha of total forest land between 1992 and 2001. In addition to this loss of total forest area, the amount of area found in smaller patches (less than 10 ha) has increased from 266,743 ha in 1992 to 356,769 ha in 2001 with the total number of patches found in these categories increasing from 297,490 in 1992 to 595,024 in 2001. Even if the single celled (one 30 x 30 m cell) patches, which can be problematic when tabulated, are ignored, the number of small forest patches more than doubled from 144,764 in 1992 to 259,134 in 2001. This increase in patches less than 5 ha in area is contrasted by a decrease in the number of patches in the middle patch size classes (10 ha to 500 ha) from 44,548 patches in 1992 to 38,062 patches in 2001 with area decreasing from 4,255,937 ha to 3,552,123 ha. The last trend to note addresses loss of the largest forest patch in 1992. The single patch of more than 25,000 ha did not remain intact resulting in increases to the numbers of forest patches in the next largest classes (1,000 ha to 10,000 ha) (Table 2.3) (Fig. 2.8).

2.3.4 Edge versus Core Forest Distinction

Between 1992 and 2001 Pennsylvania lost 335,316 ha of core forest and 180,231 ha of edge forest. This loss resulted in an increase of 475,235 ha of non-forest land cover (Table 2.4). Remembering that a portion of the core forest loss was converted to edge forest, these areas trends make sense. By examining Figure 2.10, it can be seen that changes in area can be located by examining areas for shifts in their amounts, patterns, and locations of the green and the red colors, core forest and edge forest respectively. By performing an additional GIS overlay function to specifically code for change, the result helps reveal not only the type but the locations of where the change occurred. Seventy-one percent of the core forest and 47 % of the edge forest that existed in 1992 remained in 2001. Of the core forest that changed, 7% changed to non-forest and 21% changed to edge forest. Looking at the forest area that was originally edge forest, 34% changed to non-forest cover while 19% converted to core forest. Of the 1992 non-forest areas that changed, 14% converted to edge forest and 5% converted to core forest (Table 2.5).

2.3.5 Forest Patch Change Characterization

Contingency tables were constructed to help display the forest patch size changes that occurred between the 1992 and 2001. The forest patch layers were coded in a manner that allowed tracking of the original 1992 patch class and the 2001 patch class simultaneously. The result of combining the two layers allowed for patch change evaluation and ability to track shifts among the forest patch size classes. Table 2.6 provides a more detailed view of forest patch change, which expands on that previously presented in Section 2.3.3. First, 81% of the area that was non-forest in 1992 remained in that condition in 2001. Next, 69% of the 1 ha class, 64% of the 2 ha class, 56 % of the 5 ha class and 47% of the 10 ha class converted from forest to a non-forest cover type. Not until the 25-ha patch class did more than 25% (26.2%) remain in the original forest patch size class. Larger patch classes display an increasing likelihood to remain in their original patch size class. This is partially due to their large size and the effort required when bisecting large forest areas. Most change takes place around the edges. The percent remaining in its original class climbs from 33.2% (50 ha) to as high as 69.9% for the 2,500 ha forest patch size class. The other pattern to note is that, aside from the propensity of small patch classes (2 ha through 250 ha) to shift from forest to non-forest cover, much of the area that changed class while remaining forest changed into the next smaller class. For the 500 ha class, this shift was near equal between the next smaller patch class and the next larger patch class. For the 1,000 ha class, more shifted to the next larger class. Finally, the larger patch classes shifted into smaller classes once again. In all cases, the larger percentages of the patch class shift occurred by shifting just one patch size class up or down (Fig. 2.10, Table 2.6).

2.4 Discussion

The Pennsylvania Gap Analysis Project (PAGAP) reported that different forest patterns occurred at different spatial scales (Myers et al. 2000). This fact became one of the guiding points that helped to inspire this research. Pennsylvania Gap data layers were created by identifying forest patches based on two minimum mapping units (100 ha and 2 ha) and eliminating those patches that were below each threshold. Depending on how the 100 ha minimum mapping unit (MMU) version was interpreted, 65% of the naturalistic cover in Pennsylvania could be joined into one connected patch. While, at the same time, large areas extending out from the metropolitan areas of Pittsburgh, Harrisburg, Philadelphia, and Wilkes-Barre-Scranton as well as the more open agricultural areas in the Ridge and Valley, Great Valley, and Piedmont all appear to be devoid of forest. This is contrasted when assessing the 2 ha MMU version of the forest data. Here, all of the smaller forest patches become apparent and appear as a speckled in suburban, urban, and agricultural areas and as small forest clearing within forest areas.

The validation of the two land covers shows that the land cover is reliable at distinguishing forested and non-forested areas. Achieving or, at least nearly achieving the 85% accuracy goal for both land cover interpretations described by Anderson et al.

(1976), and Congalton and Green (1999) improves the reliability of subsequent classifications. However, Congalton and Green (1999) also pointed out that Anderson et al. (1976) had no true basis for the 85% goal other than that it was sufficient and achievable. As described by Verbyla and Hammond (1995), Hammond and Verbyla (1996), and Loveland and Shaw (1996) the difficulties in achieving high accuracies on land cover interpretations while adding detailed cover types, were evident in these two land cover layers as well. As reported in the tables in Appendix B, accuracy rates dropped as generalized cover types were split to capture increased detail.

The validation process helps increase reliability of the detection of change by supporting the initial classifications. Also, focusing on forest vs. non-forest distinction helps to ground this study on the primary natural habitat of Pennsylvania. Based on the known natural history of Pennsylvania, most of the state was forested prior to European settlement (Rhoads & Block 2005). This establishes the base condition from which all departures from a forested condition represent a disturbed state and that as forest fragmentation increases and connectivity decreases the ecological integrity of natural habitat is reduced. The other issue here is that the majority of the disturbances recorded between 1992 and 2001 are not natural or temporary. As forests are converted to agricultural and urban uses, they rarely are allowed to return to their natural state. Thus, initial species compositions impacted and, at times, removed are prevented from re-colonization.

The analyses conducted here have established that forest vs. non-forest change is detectable between the 1992 and 2001 land covers. The initial, relatively simple, test looked at forest composition in the regularly spaced sampling grid used by the Pennsylvania Breeding Bird Atlas project (Brauning 1992). Here, forest percent was calculated within each cell and then differenced to reveal the change in percent forest between the two data eras. The resulting map (Fig. 2.5) illustrates that areas north and northwest of Philadelphia, southeast of Pittsburgh, and three counties along the New York border all seem to be losing forest faster than other regions of the state. While areas in north-central Pennsylvania, extending west to the Allegheny National Forest, seem to be gaining forest.

The cumulative distribution function (CDF) graphs indicate that a large percent of forested area is in smaller forest patches and that the area captured in these smaller patch sizes is increasing. Based on the two forest GIS data layers this is the first quantitative evidence that forests of Pennsylvania are fragmented and that fragmentation appears to be increasing.

As previously mentioned, Goodrich et al. (2002) reported that 57% of the forested area of Pennsylvania would be considered edge forest or forest within the first 100 m into the forest from a disturbed land cover type. That percentage was based on 1992 land cover, and methods used then were applied to capture the edge forest for the 2001 land cover. With another look at Tables 2.4 and 2.5 one can see that while 71% of the 1992 core forest remained core forest in 2001, the 29% loss amounted to over 335,000 ha of core forest. Edge forest was the most likely class to change, losing 47% of its original area. Following that trend, the 21% of the core forest that converted to edge forest would then be the most likely to convert to non-forest cover over the next nine years. When reviewing maps, the same areas near Pittsburgh, Philadelphia and along the New York border are losing core forest while gaining edge forest, although some previously cleared areas in the north-central seem to be regenerating.

In moving through the analyses, different ways of characterizing forest change become evident, thus answering the question of whether change can be characterized. The first was the changes between core and edge forest and now the next is based on forest patch size. Table 2.5 and figure 2.11 both show the same trends in patch size conversion. First, the smallest patch sizes are very likely to convert to a non-forest condition. It is not until the 100-ha patch size that the likelihood of staying in the original patch size class is higher than converting to non-forest. It follows that the larger patch sizes seem to be more cohesive. This is, in part, due to the energy required to split extensively forested areas into smaller patches. Larger areas help to maintain that condition. Also, much of the large forest patches are in public ownership and their stewardship controls new access. The larger patch sizes are also farther from large urban centers, thus not as threatened by the spreading suburban and urban fringe. An important trend to note refers to the trend showing an increase in the number of small patches between 1992 and 2001. This is particularly problematic when coupled with the fact that > 50% of the area of small patch size classes (1 ha through 25 ha) present in 1992 were converted to non-forest by 2001.

In summary, until around 1990 the use of geographic information systems and remote sensing for ecologically based studies was still not common even though both technologies were more than 15 years old (Roughgarden et al. 1991). The nationally based Gap Analysis Program began to change this in becoming the first continental habitat assessment initiative based primarily on remotely sensed land cover data (Scott et al. 1993; Loveland & Shaw 1996). Due to its reliance on land cover data the Gap program assumed a major role in helping to acquire a nationally consistent land cover interpretation now called the National Land Cover Dataset (NLCD). The second NLCD was just released in January of 2007. The land cover created for the Pennsylvania Gap Project (Myers et al. 2000) helped to define potential habitats for all of the vertebrate species. A second land cover interpretation of Pennsylvania was completed based on 2001 data. To better understand habitat condition, several state agencies and organizations helped to reprocess the original PAGAP habitat models based on these new data. This forest versus non-forest change study is the first to assess the changes in forest habitat found between the two dates.

My study has characterized change in terms of change from interior core forest to edge forest and edge forest to non-forest, and has shown that more than 50% of the edge forest in 1992 had changed to non-forest in 2001. Change also was described in the shift between various forest patch size classes. Pennsylvania's largest patch of forest (25,000 ha) in 1992 was not intact in 2001 and large percentages of the smaller patch sizes found in 1992 converted to non-forest in 2001. These two analyses point to an increase in forest fragmentation and decrease in forest connectivity.

My focus on the patch size scale as grain and extent has important implications as to how results could be applied to habitat management decisions. There is considerable research that establishes the habitat area needs of wildlife species (Merritt 1987; Fergus 2000; ELI 2003). These analyses can help locate patches of a determined minimum forest habitat, group these patches in a regional context, predict the likelihood of change based on local patch characteristics, and enhance forest patch information by adding factors such as core and edge forest distinctions, thus, by collecting these factors land managers can better target their efforts.

Table 2.1 - Land cover classification comparisons between the two land cover eras (1992 and 2001) used in this study and the National Land Cover Data (NLCD) based on Anderson et al. (1976). The study class columns represent which study class each land cover class was assigned (*Naturalistic* or *Humanistic*).

Land Cover 1992	Study 1992	NLCD - Anderson 2	NLCD - Anderson 1	Land Cover 2001	Study 2001
Water	Water	**11 Open Water** **12 Perennial Ice/Snow**	**10 Water**	Water	Water
Suburban Urban Class Urban Class	Humanistic Humanistic Humanistic	**21 Low Intensity Residential** **22 High Intensity Residential** **23 Commercial/Industry/Transportation**	**20 Developed**	Low Density Urban High Density Urban High Density Urban	Humanistic Humanistic Humanistic
Barren Transitional	Humanistic Humanistic Humanistic	**31 Bare Rock/Sand/Clay** **32 Quarrie/Strip Mine/Gravel Pit** **33 Transitional**	**30 Barren**	Beach Quarries/Coal Transitional	Humanistic Humanistic Humanistic
Deciduous Forest Evergreen Forest Mixed Forest	Naturalistic Naturalistic Naturalistic	**41 Deciduous Forest** **42 Evergreen Forest** **43 Mixed Forest**	**40 Forested Upland**	Deciduous Forest Evergreen Forest Mixed Forest	Naturalistic Naturalistic Naturalistic
Transitional	Humanistic	**51 Shrubland**	**50 Shrubland**	Transitional	Humanistic
		61 Orchards/Vineyards/Other	**60 Non-natural Woody**		
Perennial Herb	Humanistic	**71 Grasslands/Herbaceous**	**70 Herbaceous Upland**	Hay/Pasture	Humanistic
Perennial Herb Annual Herb Suburban	Humanistic Humanistic Humanistic	**81 Pasture/Hay** **82 Row Crops** **83 Small Grains** **84 Fallow** **85 Urban/Recreational Grass**	**80 Herbaceous Cultivated**	Hay/Pasture Row Crops Prob. Row Crops Low Density Urban	Humanistic Humanistic Humanistic Humanistic
Ancillary - NWI Ancillary - NWI	Naturalistic Humanistic	**91 Woody Wetlands** **92 Emergent Wetlands**	**90 Wetlands**	Woody Wetland Emergent Wetland	Naturalistic Humanistic

Table 2.2 - Validation of forest vs. non-forest classification for the two land cover layers (1992 and 2001). For the 1992 land cover, reference was videography and for the 2001 land cover reference was digital aerial photography available for the eastern third of Pennsylvania. Mapped data, in both instances, is the satellite derived land cover data. Both validations reflect accuracies at, or near, the 85% accuracy levels established as acceptable by Anderson et al. (1976) and Congalton and Green (1999).

1992 Forest vs. Non-Forest Validation

Nat/Hum	Rwater	Rhuman	Rnatural	Count	Percent
Mwater	40.0	5.0	10.0	55.0	72.7
Mhuman	3.0	170.0	65.0	238.0	71.4
Mnatural	4.0	40.0	418.0	462.0	90.5
Count	47.0	215.0	493.0	**755.0**	
Percent	85.1	79.1	84.8		83.2

**Table summarized from the PA GAP report (Myers et.al 2000).

2001 Forest vs. Non-Forest Validation

Nat/Hum	Rwater	Rhuman	Rnatural	Count	Percent
Mwater	18.0	2.0	1.0	21.0	85.7
Mhuman	0.0	60.0	13.0	73.0	82.2
Mnatural	2.0	7.0	103.0	112.0	92.0
Count	20.0	69.0	117.0	**206.0**	
Percent	90.0	87.0	88.0		87.9

R – refers to reference data
M – refers to mapped data

Table 2.3 - Forest patch distribution for each land cover era, 1992 and 2001. Patch size class, the number of patches in that class, total area (ha) in that patch size class, and percent of Pennsylvania forest found in that patch size class are totaled.

Patch Size Class (ha)	**1992 Land Cover Patches**	**Area(ha)**	**Area %**	**2001 Land Cover Patches**	**Area(ha)**	**Area %**
***Single Cells**	152,726	13,745.34	0.19	335,890	30,230.10	0.46
1	103,700	63,038.43	0.89	214,168	125,612.64	1.91
2	26,194	84,100.14	1.19	29,960	94,637.97	1.44
5	14,870	105,859.26	1.49	15,006	106,288.92	1.62
10	14,848	237,591.54	3.35	13,414	213,926.94	3.25
25	9,046	324,483.21	4.58	7,933	283,788.99	4.32
50	8,204	589,938.03	8.32	6,744	483,679.80	7.36
100	8,421	1,326,302.82	18.71	6,581	1,031,629.95	15.69
250	2,892	992,600.55	14.00	2,310	795,600.72	12.10
500	1,137	785,020.50	11.07	1,080	743,496.30	11.31
1,000	701	1,084,773.06	15.30	711	1,092,835.62	16.62
2,500	219	760,756.50	10.73	246	866,734.02	13.18
5,000	79	513,177.93	7.24	80	519,050.16	7.90
10,000	14	182,181.42	2.57	15	186,694.02	2.84
25,000	1	26,900.55	0.38	0	0.00	0.00
Totals	**343,052**	**7,090,469.28**		**634,138**	**6,574,206.15**	

Table 2.4 - Non-forest, edge forest and core forest for each land cover era. The last column indicates the net loss or gain of forest area (ha) in each class that occurred between 1992 and 2001.

HA_1992	CLASS	HA_2001	Net Loss/Gain
4515489.09	Non-Forest	4990723.91	475234.82
3147230.88	Edge Forest	2966999.83	-180231.05
3943238.40	Core Forest	3607922.47	-335315.93

Table 2.5 - Edge versus Core Forest change that occurred between 1992 and 2001. The first three rows show the percentage of each forest type that remained in the same type class (i.e., 71 % of the Core Forest in 1992 remained Core Forest in 2001). The next six rows break down the types of forest conversions that occurred.

FOREST CHANGE

Same Forest Type	Percent
Non-Forest	80.82
Edge Forest	47.09
Core Forest	71.43
Change in Forest Type	**Percent Change**
Non-Forest to Edge	14.38
Non-Forest to Core	4.79
Edge to Non-Forest	34.23
Edge to Core	18.67
Core to Non-Forest	7.19
Core to Edge	21.38

Table 2.6 - Percent forest patch size change that occurred between 1992 and 2001. When viewing the 10 ha column, 21.8% of the area that was 10 ha patch size class in 1992, remained in that class in 2001. Then, 8.2% of that area decreased into the 5 ha patch class and 5.2% increased into the 25 ha patch class. Note, that 46.5% of the original area of the 10 ha patch size class, in 1992, was converted to non-forest by 2001.

Patch Class	NotForest	1 ha	2 ha	5 ha	10 ha	25 ha	50 ha	100 ha	250 ha	500 ha	1000 ha	2500 ha	5000 ha	10,000 ha	25,000 ha
NotForest	81.11	69.12	64.18	56.15	46.51	38.20	31.68	25.16	19.26	12.35	8.36	6.45	6.38	4.85	5.92
1 ha	1.87	6.40	7.89	5.19	3.36	2.16	1.51	0.99	0.69	0.39	0.25	0.18	0.17	0.12	0.20
2 ha	0.76	2.72	9.60	8.48	3.86	2.00	1.17	0.66	0.40	0.22	0.14	0.10	0.11	0.09	0.22
5 ha	0.66	1.35	3.98	12.67	8.16	3.06	1.57	0.78	0.41	0.21	0.15	0.12	0.11	0.13	0.29
10 ha	1.01	1.62	2.19	5.13	21.79	13.01	4.38	1.68	0.76	0.38	0.24	0.22	0.16	0.31	0.56
25 ha	1.08	1.49	1.62	2.13	5.18	26.24	12.95	2.74	0.89	0.40	0.28	0.26	0.20	0.36	0.47
50 ha	1.52	1.96	1.59	2.02	2.77	6.49	33.23	11.35	1.92	0.72	0.42	0.27	0.26	0.51	0.50
100 ha	2.77	3.39	2.47	2.71	3.09	3.91	8.38	47.38	15.77	2.14	1.21	0.86	0.66	0.62	1.00
250 ha	1.94	2.36	1.68	1.59	1.64	1.82	2.07	5.59	48.51	11.73	1.84	0.69	0.80	0.76	0.75
500 ha	1.78	2.32	1.51	1.20	1.18	1.08	1.10	1.91	7.52	55.60	7.57	1.48	1.45	2.79	5.02
1000 ha	2.32	3.12	1.44	1.29	1.18	1.02	1.06	1.03	2.55	11.82	67.06	10.15	2.86	6.48	17.32
2500 ha	1.70	2.26	1.03	0.78	0.74	0.66	0.65	0.41	0.94	3.17	9.99	69.88	14.84	11.39	0.01
5000 ha	1.15	1.51	0.70	0.57	0.44	0.27	0.20	0.28	0.31	0.45	1.64	7.93	64.18	19.06	29.33
10,000 ha	0.32	0.40	0.12	0.09	0.08	0.08	0.07	0.04	0.08	0.41	0.83	1.44	7.82	52.51	38.43

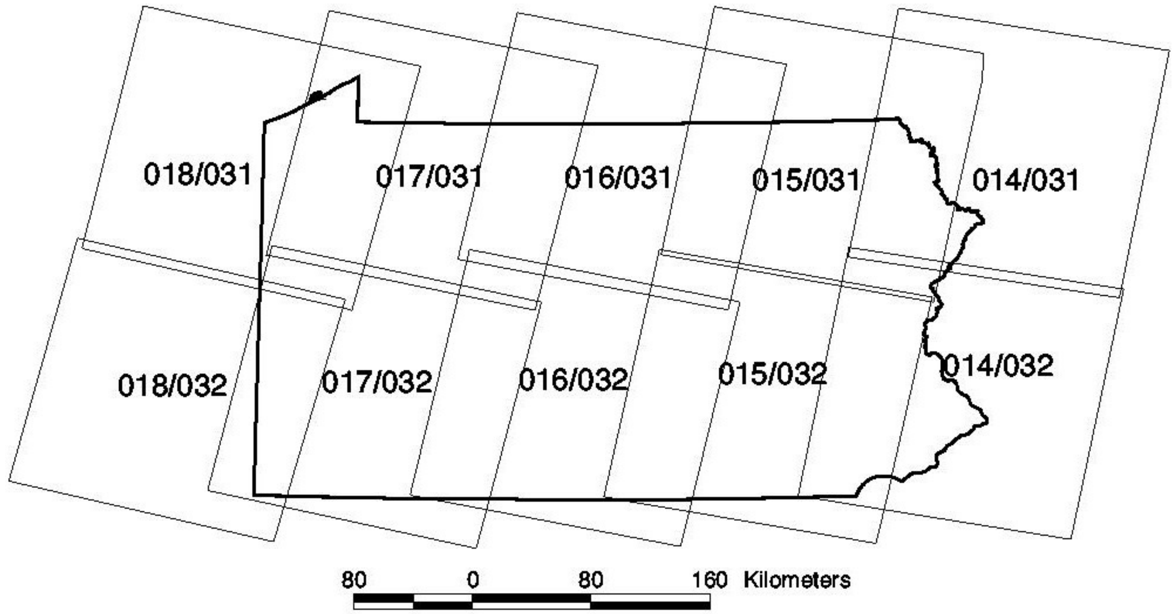


Figure 2.1 - Landsat TM and ETM coverage of Pennsylvania by path-row position. See Appendix A for a list of imagery dates for each era (1992 and 2001) and metadata documents for each land cover data layer.

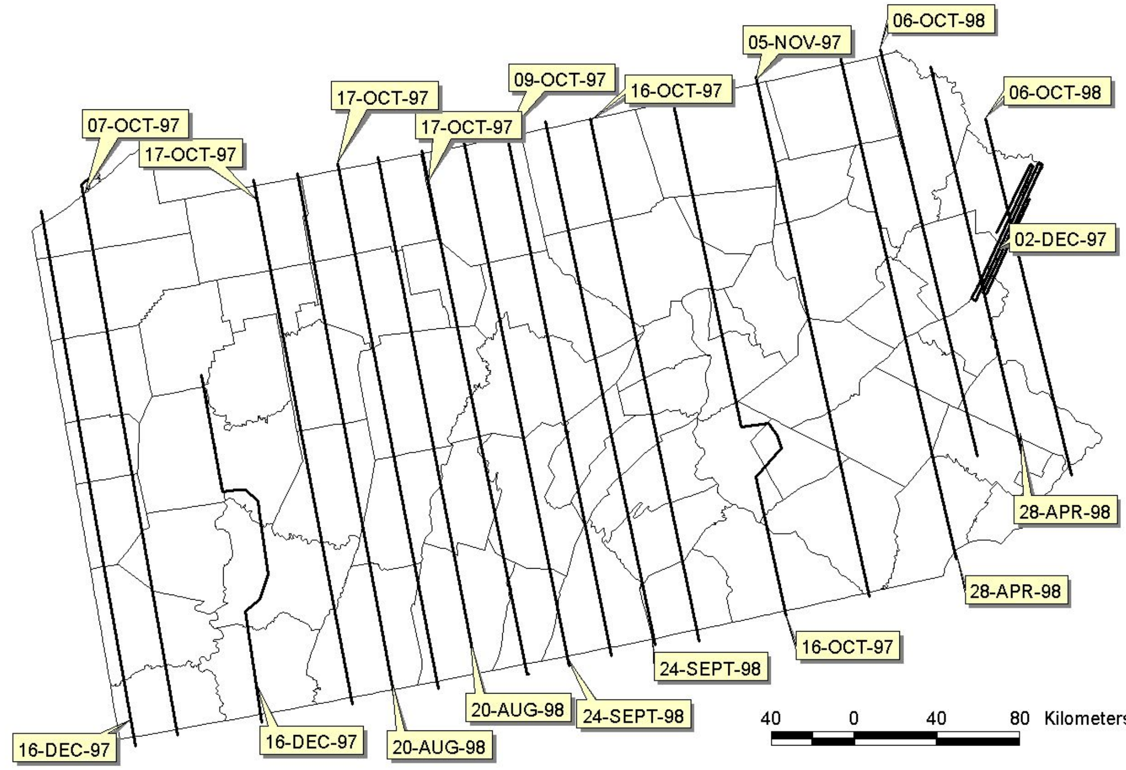


Figure 2.2 - Aerial videography transects used for validation of 1992 land cover. Transects in the eastern third of Pennsylvania (six transects) also were used for validation of the 2001 land cover. Random points were placed along each transect and then validated with videography, for 1992 data, or with digital aerial photography, for 2001. Dates indicate when the original videography was collected.

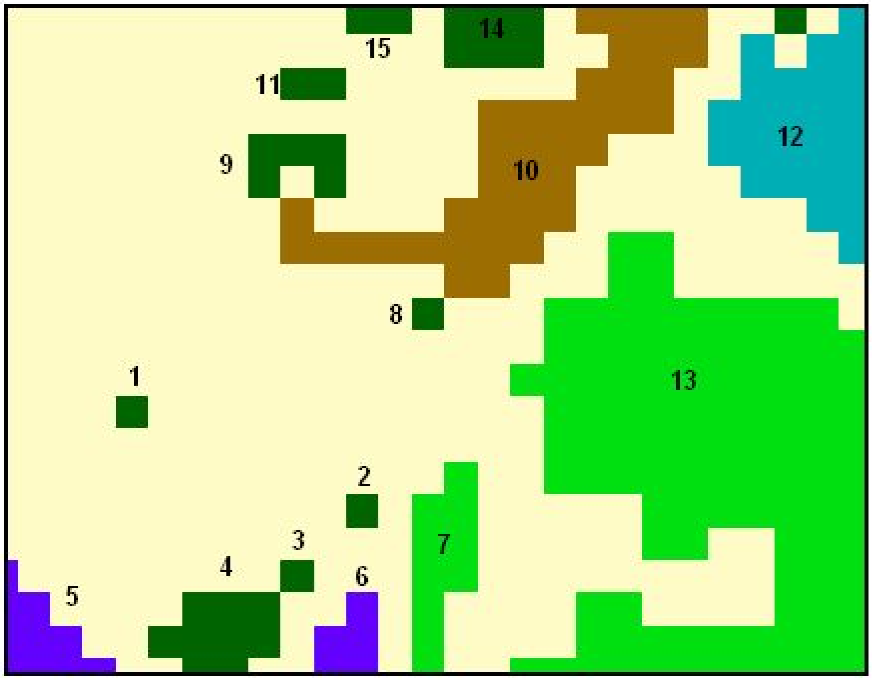


Figure 2.3 - Cell configuration for a subset of forest patches. For a single cell to be included as part of an existing patch the cell and the patch must share a cell edge. Isolated cells and cells that touch solely at corners were considered separate patches. For example, patches 1, 2, 3, and 8 are single cell patches.

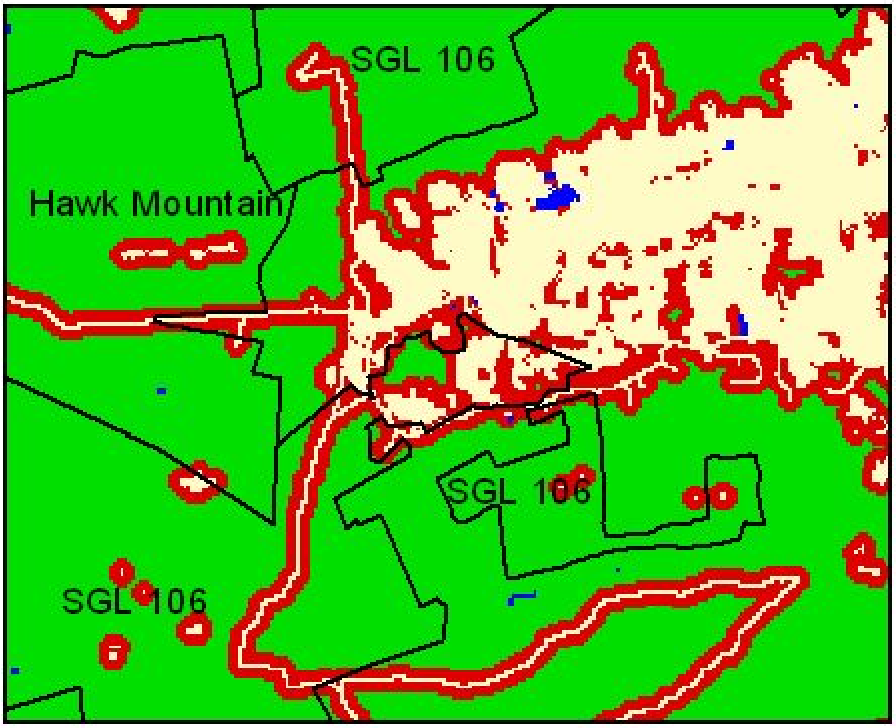


Figure 2.4 - Close-up of the Hawk Mountain area located 50 km north of the city of Reading in Berks County Pennsylvania illustrates core versus edge forest. Green represents core forest, pale yellow is suburban or agriculture, blue is water, and red represents forest within the first 100 m of a non-forest land cover.

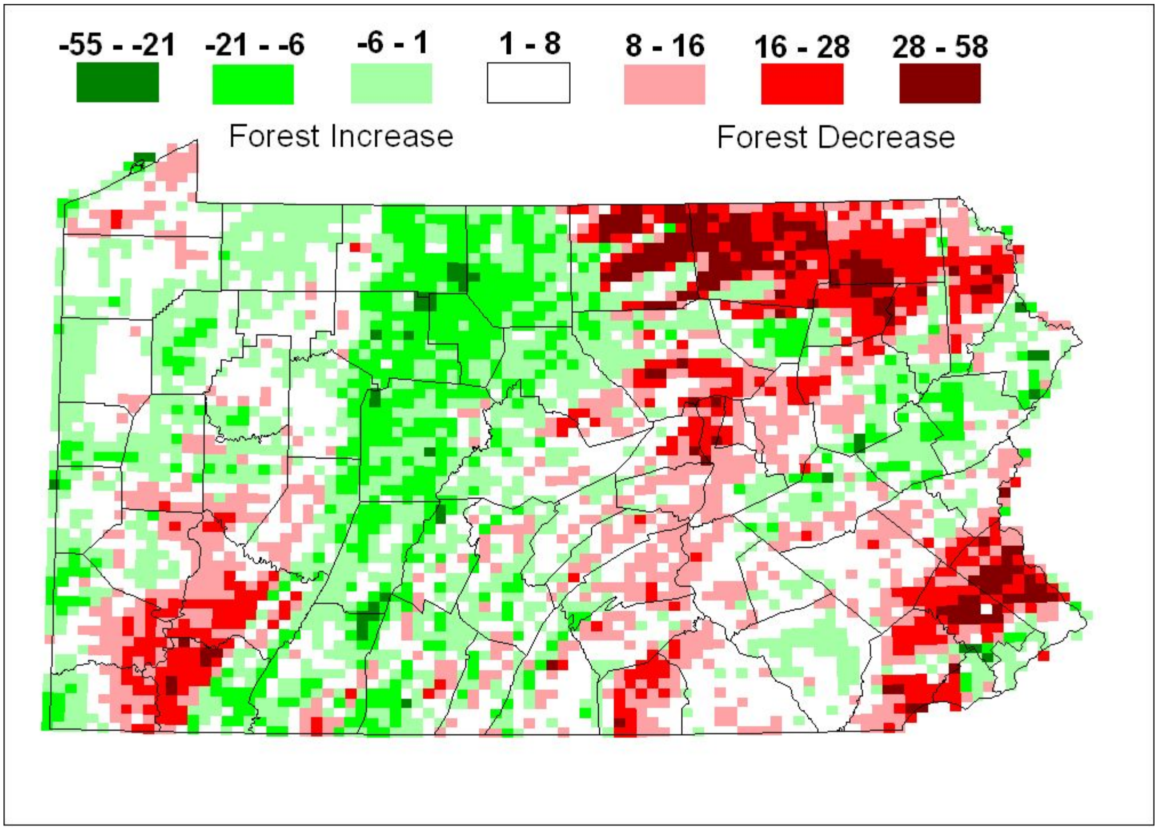


Figure 2.5 – Depiction of percent forest change as calculated within the Pennsylvania Breeding Bird Atlas (BBA) sampling blocks. Percent forest was calculated for each land cover era (1992 and 2001) within each block and then differenced. Each cell is approximately 24.3 km^2. Numbers across the top indicate percent forest change.

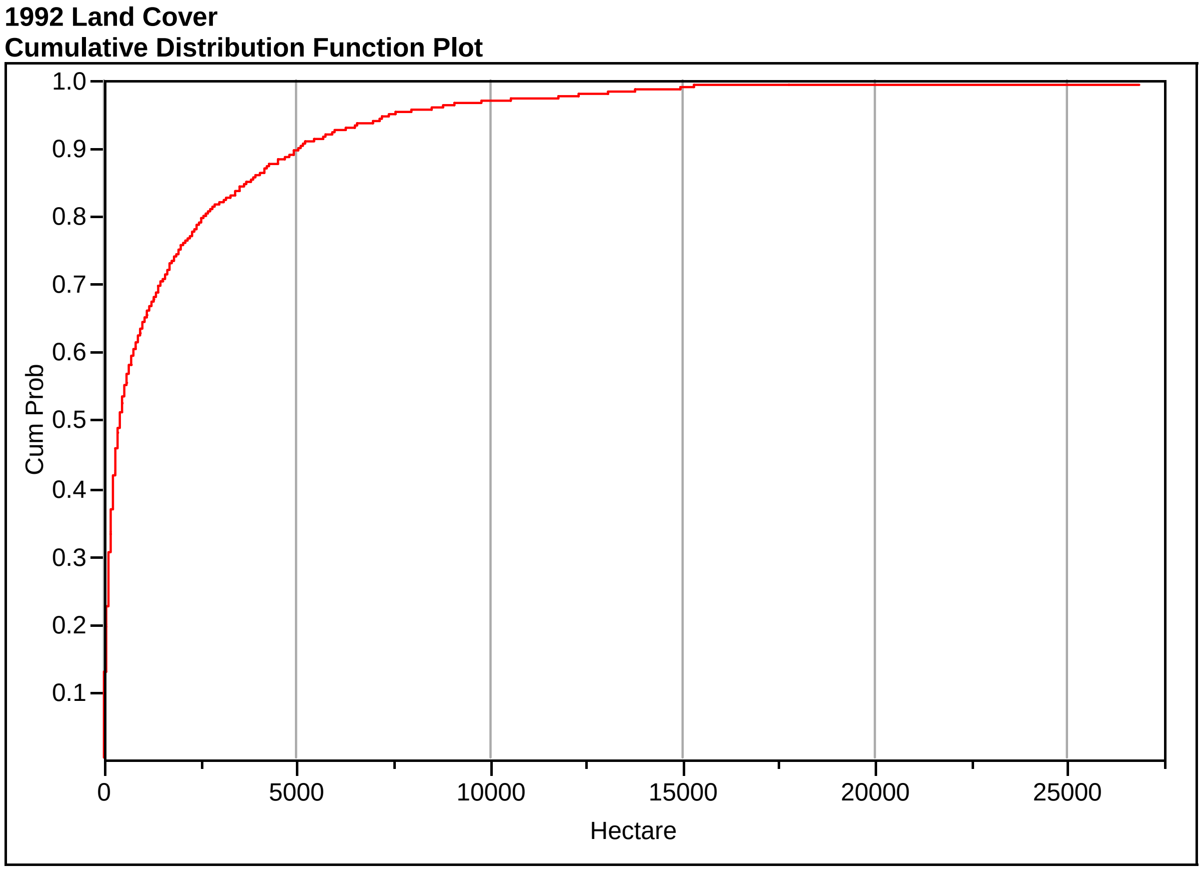


Figure 2.6 - Forest patch size for the 1992 land cover layer represented as a general distribution in a Cumulative Distribution Function (CDF) plot. Forest patch size represented as a continuous variable was tabulated. The graph illustrates that a large majority of the forested areas of Pennsylvania occur in small patches.

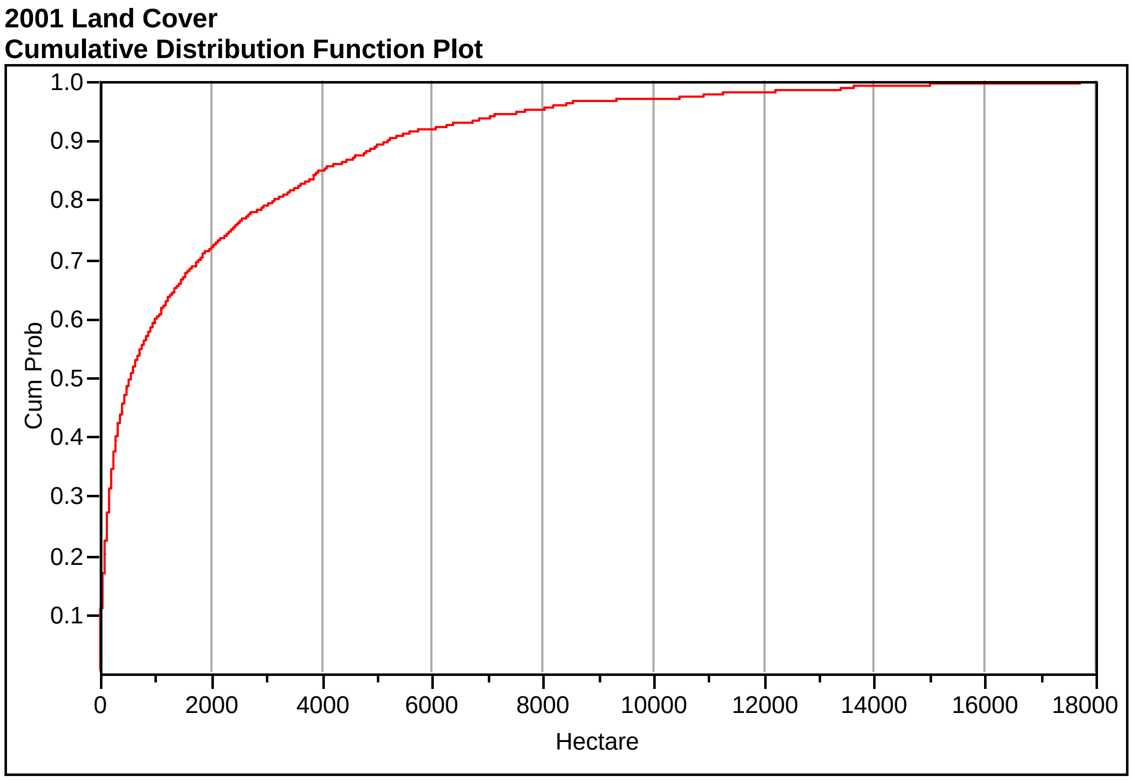


Figure 2.7 - Forest patch size for the 2001 land cover layer represented as a general distribution in a Cumulative Distribution Function (CDF) plot. Forest patch size represented as a continuous variable was tabulated. The graph illustrates that a large majority of the forested areas of Pennsylvania occur in small patches.

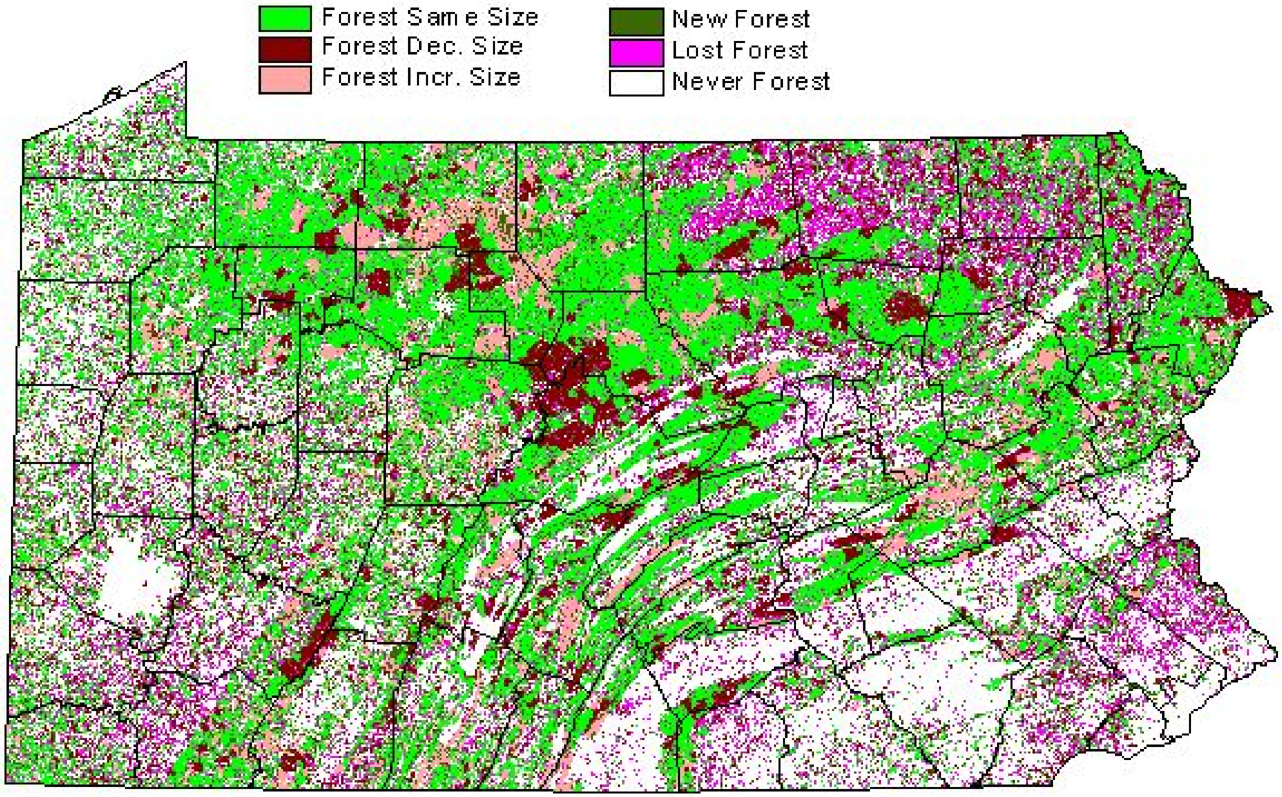


Figure 2.8 – Forest patch class change between land cover eras. Solid colors represent patch size change (burgundy increase and salmon decrease). Green, visibly the majority of forest area, did not change patch size class. Pink represents areas that lost forest between 1992 and 2001.

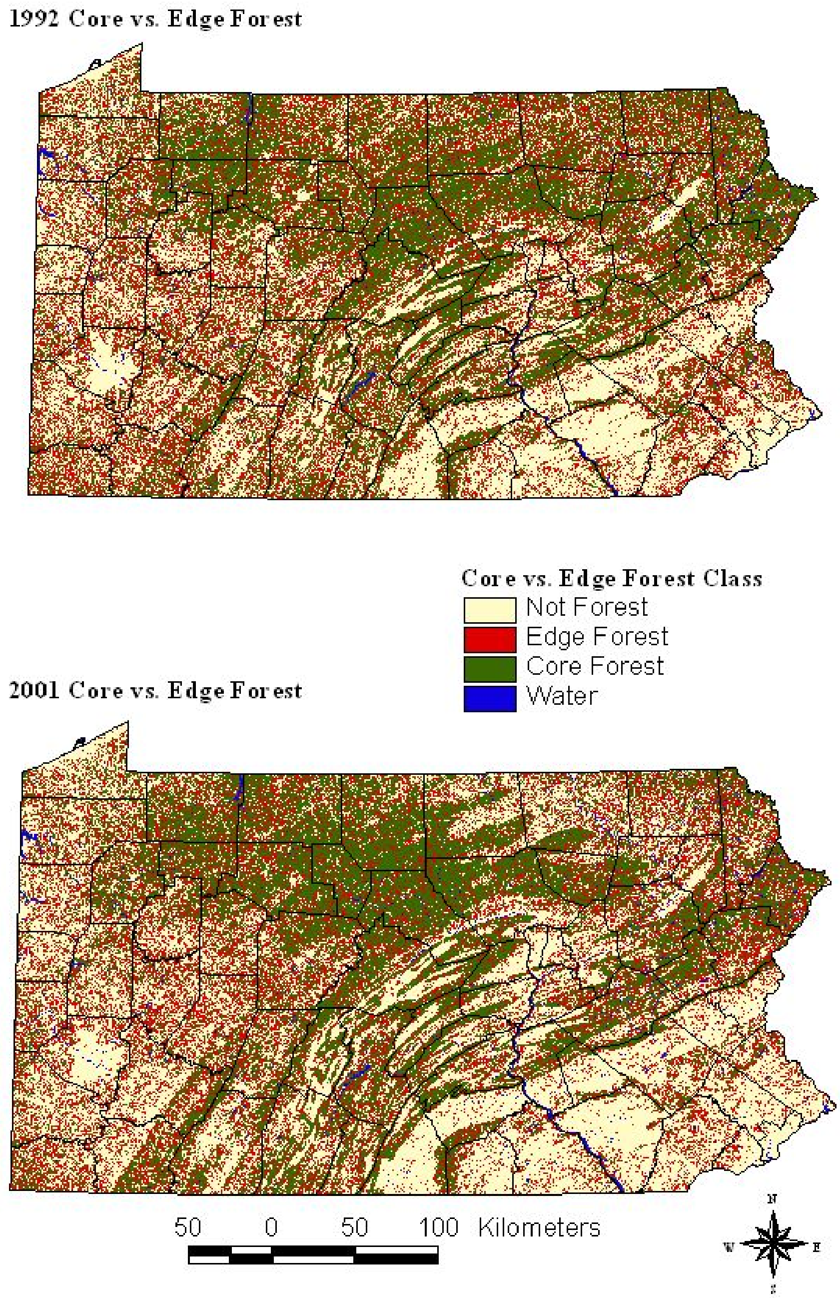


Figure 2.9 - Edge versus core forest for each era. The red color "Edge Forest" represents forested areas within the first 100 m into forest cover. Several areas appear to have increased in their core forest between 1992 and 2001 (i.e., became more green). Several areas with high edge densities in 1992 have lost forest area and, appear clear and less red or green (i.e., non-forest). Three regions to note: 1) areas extending north, and northwest from Philadelphia into the Great Valley, 2) southeast of Pittsburgh extending to the Laurel Highlands, and 3) northeastern counties along the border with New York.

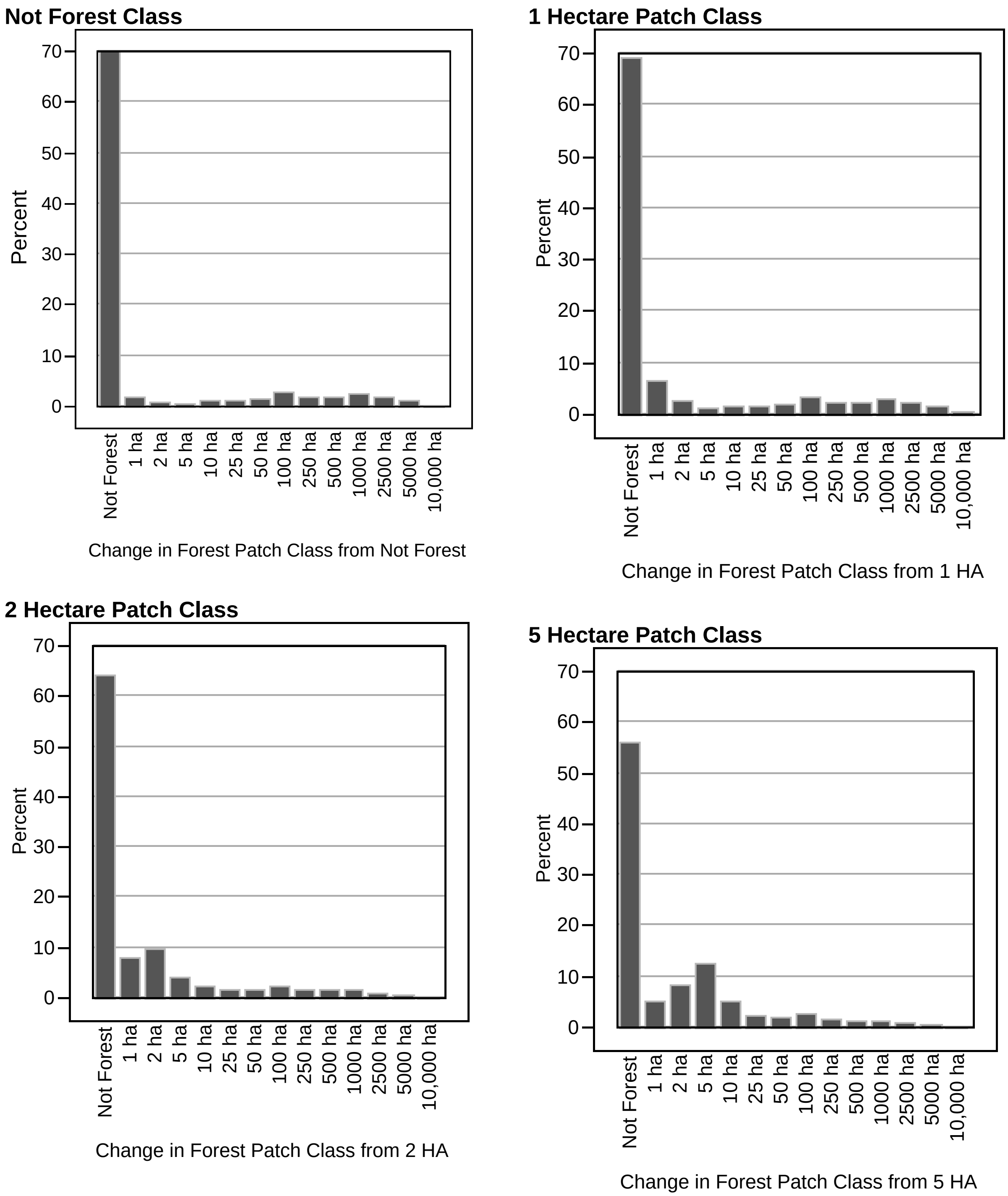


Figure 2.10a - Graph series illustrates change in forest patch size between 1992 and 2001.

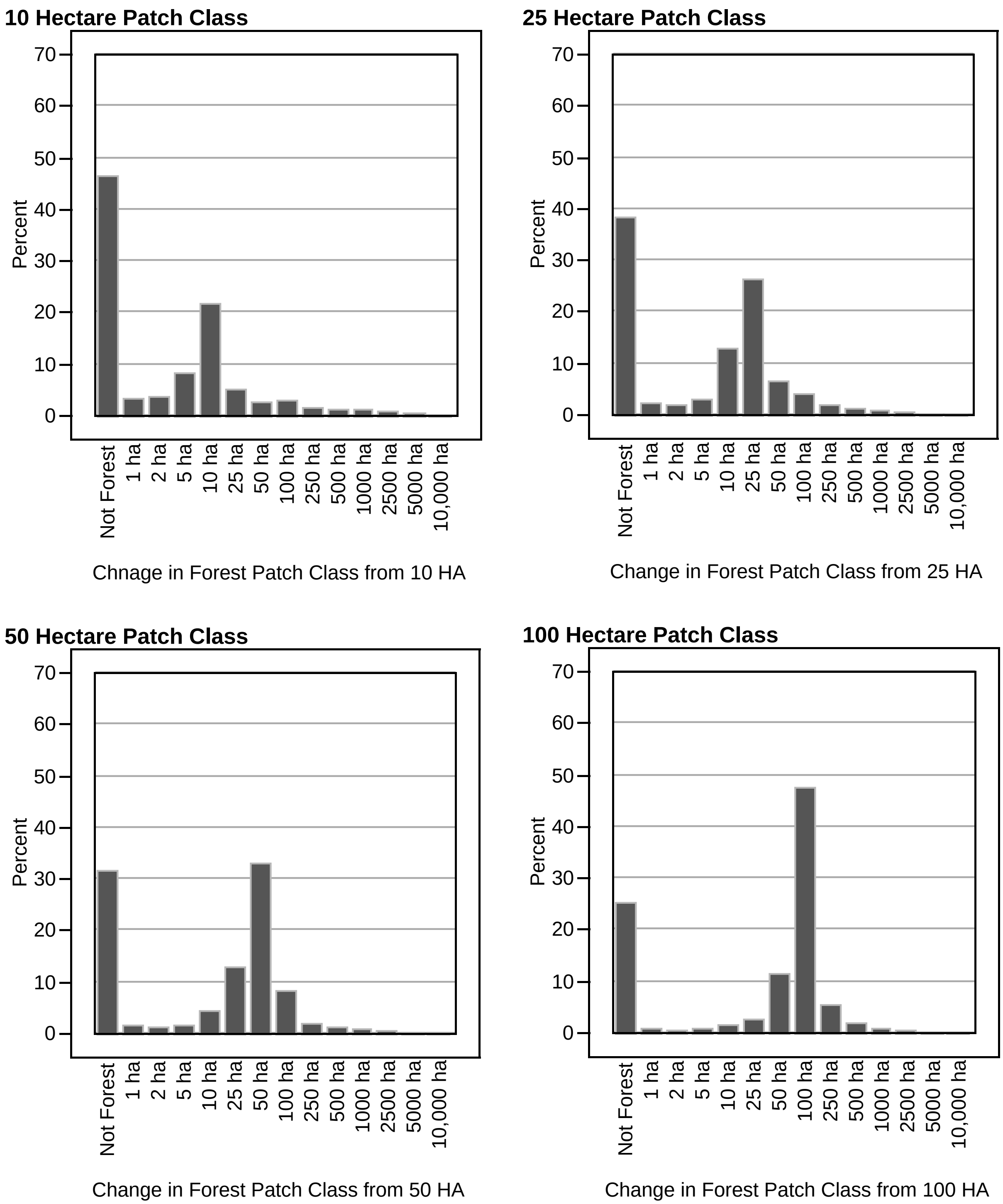


Figure 2.10b - Graph series illustrates change in forest patch size between 1992 and 2001.

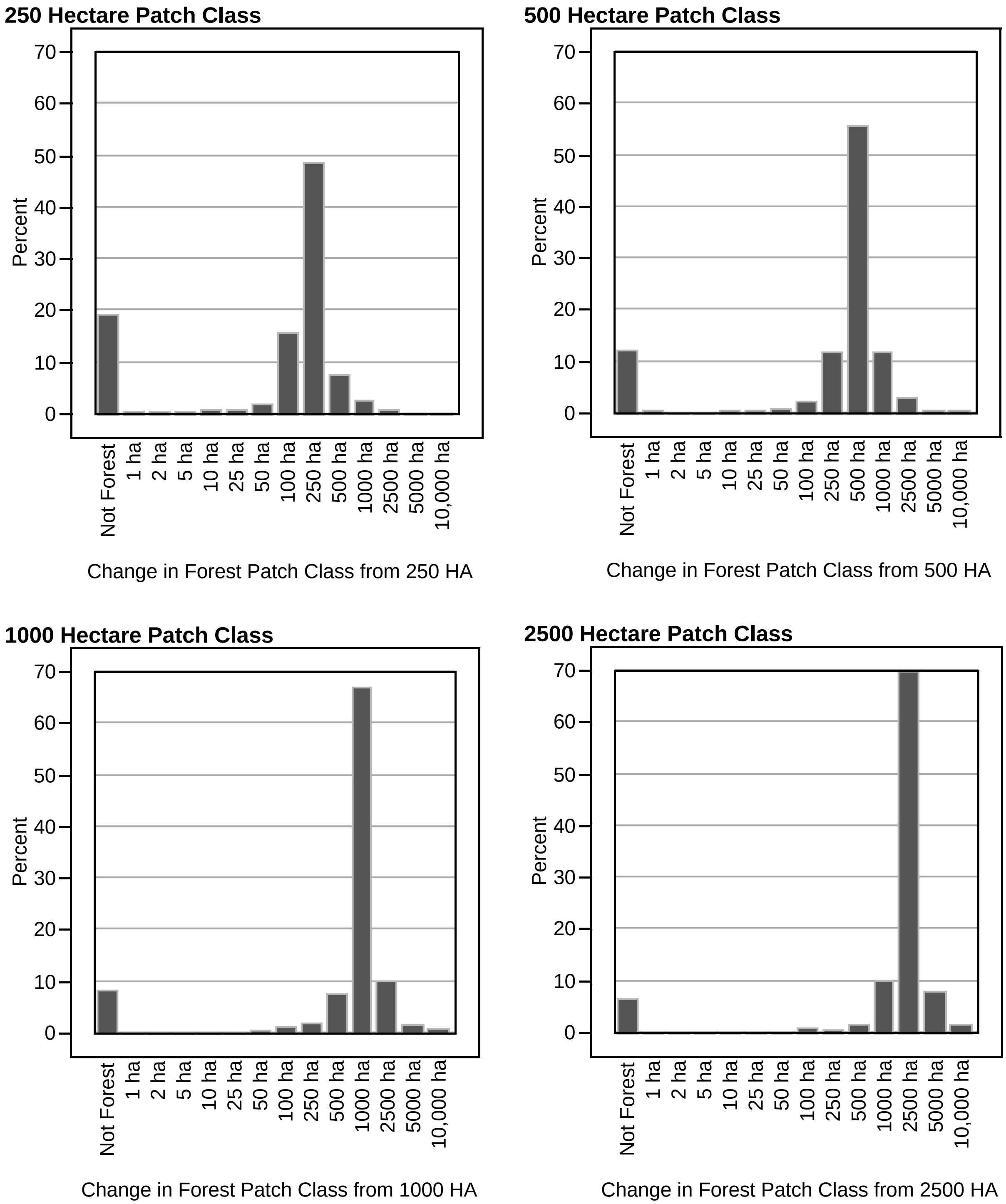
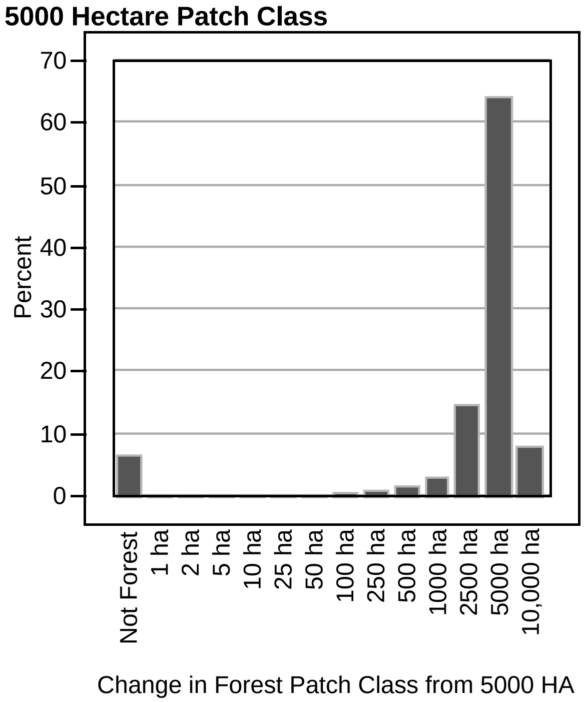
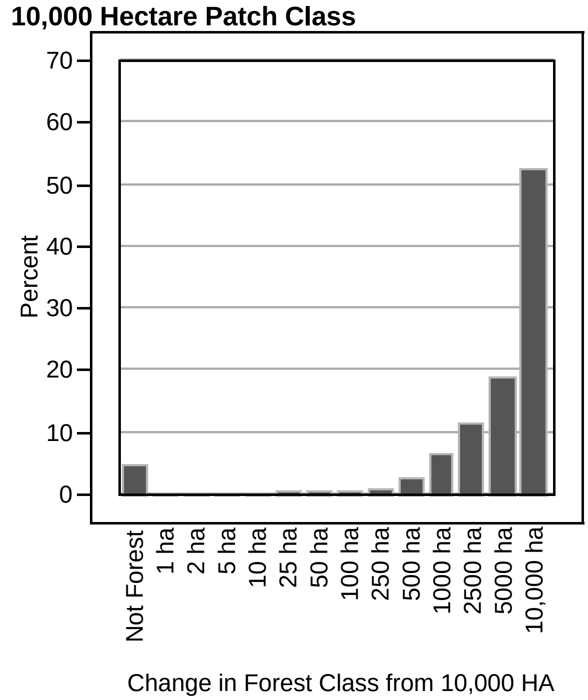


Figure 2.10c - Graph series illustrates change in forest patch size between 1992 and 2001.





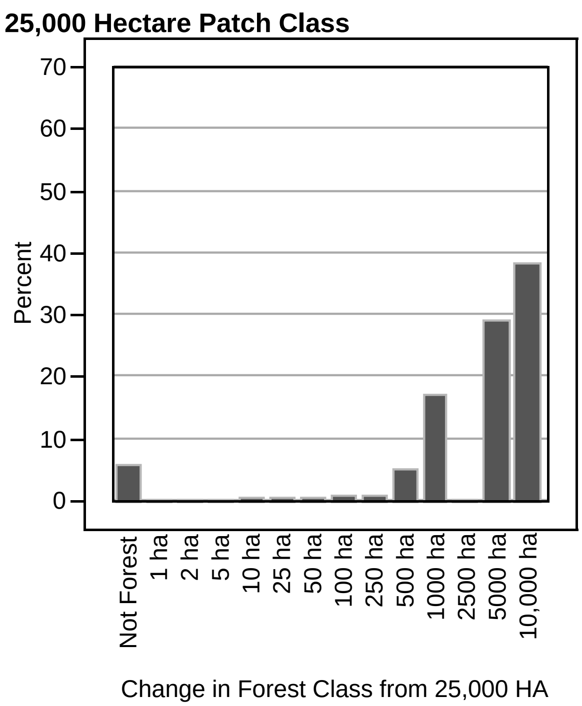
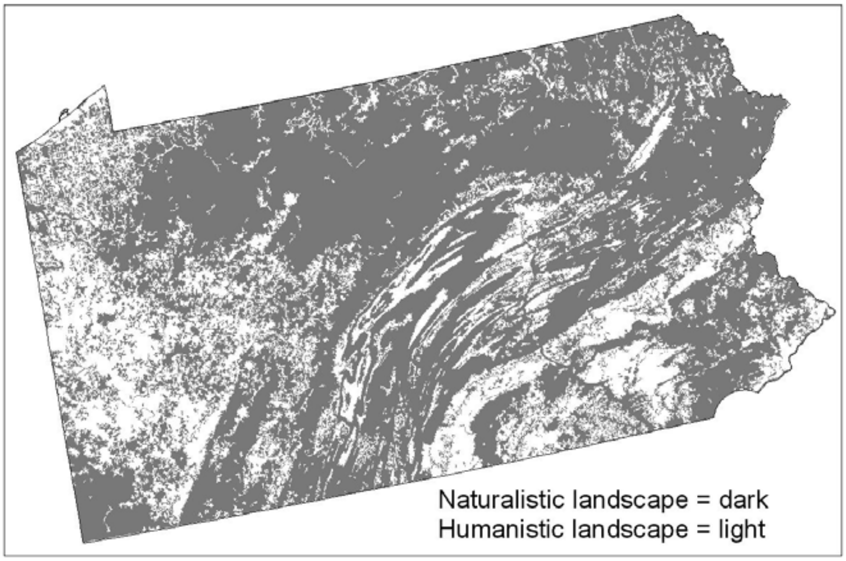


Figure 2.10 - Graph series illustrates change in forest patch size class found between the two land cover eras 1992 and 2001. The X-axis indicates the percent of the original 1992 forest class area that either stayed in the same class or shifted into a new class in 2001. Note that there was not a 25,000 ha class in the 2001 land cover layer.

Minimum mapping unit – 100 ha



Minimum mapping unit – 2 ha

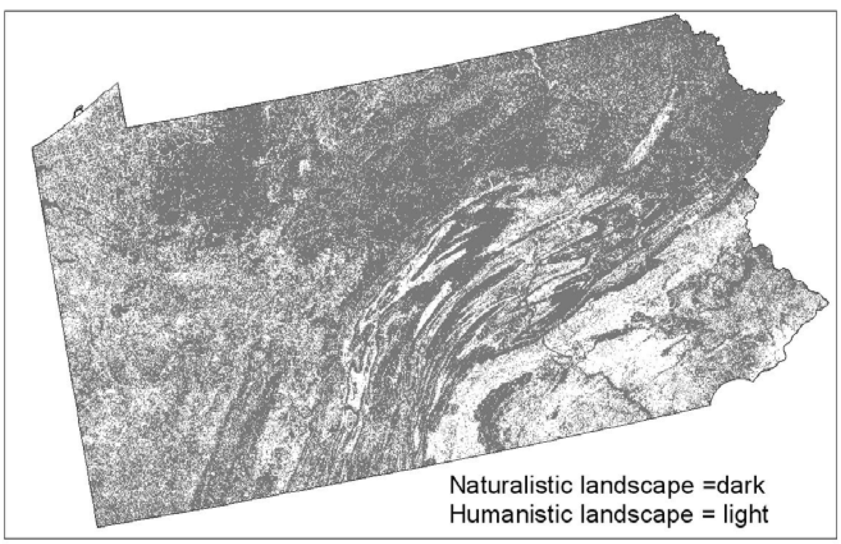


Figure 2.11 - Based on the 1992 land cover these two maps show forest cover mapped at two different minimum mapping units (MMU), 100 ha and 2 ha. In the 100 ha rendering it is easy to see the larger white areas and the more solid gray areas. The gray speckled areas, in the 2 ha MMU map, are the forest patches that were dissolved out because they were too small and the more solid gray areas depict the small forest clearings that were treated as forest.

Chapter 3

Landscape Characteristics of Forest Fragmentation Change as Captured by Ecological Landtype Associations

3.1 Introduction

3.1.1 Land Classification and Ecological Units

For purposes of resource management and ecological assessment, global land surfaces have been classified and mapped since the ancient Greeks (Bailey 1996). Baron von Humboldt created an outline for world plant and animal relationships that incorporated climate, latitude, and elevation (Bailey 1996). Herbertson (1905) mapped the "major natural regions" of the world using temperature regimes as the primary classifier and then regional topographic conditions such as elevation and landform as modifiers. Similar classifications were completed by Merriam (1898) in the United States and Holdridge (1947) for tropical regions. This work, and others, helped establish the basis of ecosystem classifications. Rowe (1961) defined an ecosystem as "topographic unit, a volume of land and air plus organic contents extending aerially over a particular part of the earth's surface for a certain time." This definition connects earth surface features with atmospheric processes and suggests a boundary outlining the area of the topographic unit as well as an associated time element. In contrast, Bailey (1996) defines ecosystem as "an area of any size with an association of physical and biological components so organized that a change in any one component will bring about a change in the other components and the operation of the whole system." This version does not impose a size limit and emphasizes the physical and biological interdependencies especially in response to change.

Bailey (1983, 1987) and Omernik (1987), then further refined by Bailey (1995) and Omernik (1995), described and delineated ecoregions of the United States. These were defined at continental and regional scales and incorporated climatic regimes and physiographic features. The U.S. Environmental Protection Agency adopted ecoregions, as defined by Omernik (1995) and participates in their further refinement as part of the North American Commission for Environmental Cooperation (CEC) which was formed in 1997 (USEPA 1996; National Atlas 2007). The U.S. Forest Service adopted the National Hierarchical Framework of Ecological Units (ECOMAP 1993; USDA 1994; USDA 1995) classifying eight levels of ecological units starting with Bailey's ecoregions at the broad scale, and then sequentially nesting each level into finer scaled units. Both Bailey's and Omernik's ecological mapping procedures are similar and combine physical features such as climate, topography, physiography, and geology, with organic features, such as soils, vegetation, and wildlife to help delineate boundaries. The Forest Service, in cooperation with the Pennsylvania Bureau of Forestry, delineated the first five levels of units for Pennsylvania (Myers 2000; Kong 2006).

Kong (2006) delineated ecological unit level six, the Landtype Association (LTA). LTAs are groups of complementary landscape features that combine through spatial adjacency to create ecological contrasts across regions. These factors affect biotic distributions, hydrologic function, natural disturbances and land use patterns (Myers 2000). Kong (2006) based the LTA delineations on topological features that incorporated both hydrologic and habitat conditions with the assumption that these features capture ecological influences at the landscape scale.

3.1.2 Land Classification and Hydrological Units

Watersheds are another form of land surface classification used by landscape researchers and resource managers. Watersheds are particularly suited for encapsulating aquatic habitats and establishing management units to assess impacts of agricultural runoff (erosion, etc.) and other point and non-point pollution sources (USEPA 1996; Omernik & Bailey 1997; Naiman 1998). Conceptually, watersheds are easier to define than ecoregions. Watersheds are topographic areas where surface water drains to a specific point, usually on a stream or other waterbody. There are an infinite number of points that could form topographic watersheds, although streams confluences are normally used (Omernik & Bailey 1997).

Seaber et al. (1987) and Griffith et al. (1999) defined a hierarchical classification of watersheds for the United States. This U.S. Geological Survey report describes a system of nested Hydrologic Unit Coded (HUC) watersheds with each watershed level defined by local or regional drainage characteristics and minimum and maximum drainage area thresholds. The coding establishes a spatial identifier and is comprised of a series of digits. As the codes increase their number of digits the watershed area decreases and becomes finer scaled. For example, an 8-digit HUC watershed covers a larger area than a 14-digit HUC watershed. Thus, a local erosion improvement project would be conducted within a 14-digit watershed, whereas a regional assessment might be based within the area of an 8-digit watershed (Seaber et al. 1987).

Naturally formed boundaries, such as watersheds and landtypes, have been previously used as a means of encapsulating and analyzing landscape information and are suited as a method of communicating condition to resource managers and the general public. Using natural ecological boundaries helps define influences on ecological condition within each unit. This helps mitigation efforts because it is easier to identify the impacts resulting in the current condition (Bryce et al. 1999; Detenbeck et al. 2000).

Several watershed based classifications have been completed recently for the Mid-Atlantic region using geographic information system (GIS) technology and the wealth of geospatial data that are available (Jones et al. 1997). Johnson et al. (2001), used watersheds similar in size to an 8-digit HUC, to predict surface water pollution, based on land cover patterns. Jones et al. (2001) predicted nutrient and sediment loading, based on a suite of landscape metrics. Argent et al. (2002) classified watersheds based on predicted habitat suitability for fish. This study emphasized that both regional, large area watersheds, and local stream characteristics affect suitability for fish habitat. For the entire Mid-Atlantic region, Wardrop et al. (2005) classified 11-digit and 14-digit HUC watersheds based on land cover condition and topographic features and then stratified them based on ecological regions to predict human-influenced condition. Using cluster analysis their study found that, in addition to general land use, riparian land cover and topographic variability helped to group watersheds based on their current condition. For Pennsylvania and West Virginia, Myers et al. (2006) classified 14-digit HUC watersheds based on natural characteristics found in a compilation of four collections of variables: (1) physical and topographic, (2) soil and geologic, (3) climate, and (4) hydrologic. Griscom et al. (In Prep.), under the same project, used this classification to predict the vulnerability to human stressors by adding land cover characteristics and atmospheric deposition to the previous watershed classification. This identified watersheds possessing certain characteristics that made them vulnerable to changes in human stressors.

3.1.3 Landscape Fragmentation

Fragmentation and its resulting habitat loss is considered by many scientists to be the most significant threat to biodiversity today (Noss 1987a; Bennett 2003; Hilty et al. 2006). As mentioned in Chapter 2, habitat fragmentation has been described as the breakup and conversion of extensive habitats into smaller isolated habitat fragments too small to support their original species compositions (MacArthur & Wilson 1967; Wilcove 1987; Myer 1994; Fahrig 2003). Harris (1984) noted two components of fragmentation: (1) conversion of natural habitat in a landscape to other covers, and (2) separation and isolation of the remaining natural habitat into smaller patches. Forest fragmentation was identified as the most important threat to the health of natural resources of Pennsylvania (21^{st} Century Environment Commission 1998; Goodrich et al. 2002)

Landscape metrics have been created to help measure the effect of land cover configuration on landscape function (Forman & Godron 1986; Plotnick et al. 1993; McGarigal & Marks 1995; Riitters et al. 1995; Miller et al. 1997; Opdam et al. 2003; Li & Wu 2004; Neel et al. 2004). These metrics generally fall into one of three categories; (1) spatial heterogeneity metrics, (2) fragmentation metrics, and (3) edge metrics.

Spatial heterogeneity metrics evaluate variety and configuration of land covers found in a landscape. These include number and proportion of land covers, along with their distribution and arrangement. Spatial heterogeneity metrics can only be calculated for an entire defined landscape.

Fragmentation metrics focus on forest cover and reflect patch metrics related to number, size, and degree of isolation of forest patches. These include measures of forest interior, forest edge, and forest connectivity. Fragmentation metrics can be calculated for both complete landscapes as well as individual cover types.

Edge metrics evaluate the interface among land covers. These include edge densities, amounts of shared edge between individual cover types, and metrics that summarize perimeter-to-area ratios, such as fractal dimension. Edge metrics also can be calculated for both complete landscapes as well as individual cover types (Dunn et al. 1991; McGarigal & Marks 1995).

Research Objectives

- *What forest change and forest patch size change is bounded by landtype associations (LTAs)?*
- *Can change patterns be identified for LTAs that appear to be losing forest habitat more rapidly?*
- *Do LTAs group regionally based on fragmentation change intensities?*
- *Can LTAs be used to predicted vulnerability to continued change based on these characterizations?*

3.2 Methods

3.2.1 Classification of Landtype Associations

Kong (2006) based the LTA delineations on topographical features that incorporated both hydrologic and habitat conditions, with the assumption that these features capture ecological characteristics at the landscape scale. This delineation divides Pennsylvania into three physically structuring formations or landtype associations: termed highland habitat (HH), transitional terrace (TT), and dual drainage (DD).

Highland habitat contains headwater streams and much of the area over ridges and watershed divides. These features are predominately upward arching areas in the landscape. Transitional terrace areas are similar to highland habitat but in more intermediate areas of terrain, such as mountain saddles and maintain connections to HH areas via narrow land constrictions. Dual drainages contain small stream drainages or ravines, and large river drainages. These features tend to be downward dipping structures in the landscape. Landtypes then subdivide into 18 subtypes, incorporating local and regional features in their delineations (Table 3.1). Kong (2006) validated the delineations with data from stream networks, land cover patterns, and vertebrate habitats, and LTAs captured much of the variation in these data. The final LTA data layer contains 10,782 polygons classified into one of the three landtype associations and divided into 18 subtypes. This study uses the LTA delineation to characterize areas of varying forest fragmentation to identify regions displaying similar rates of fragmentation change. Using the procedures described below, measures of change for a series of forest patch sizes and landscape metrics were calculated for each LTA boundary.

The area of LTAs makes them particularly useful for this study. As stated before, LTA delineations encapsulate landscape features affecting biotic distributions, hydrologic function, natural disturbances, and land use patterns. Their area, mean area in Pennsylvania = 1078 ha, is small enough to capture local forest patterns while large enough to eliminate known computational errors encountered when tabulating landscape metrics in smaller areas.

3.2.2 Landscape Metrics & FRAGSTATS calculations

Using two forest data layers (see Chapter 2), composition of each forest patch size class and several landscape metrics were calculated using a specifically programmed ArcView 3.3 (ESRI 2002) project that automates the processing of multiple polygons (Bishop & Lehning 2007). The computer program incorporates landscape metrics included with the Patch Analyst extension to ArcView (Rempel 2007) with several more standard geographic information system (GIS) functions that calculate landscape metrics

(Brooks et al. 2004; Rheinhardt et al. 2006; Wentworth 2006). Patch Analyst software includes all of landscape metrics programmed for the software FRAGSTATS (McGarigal & Marks 1995) and was designed to be compatible with ArcView (3.X) GIS software. This software was used to calculate landscape metrics for this study.

Composition of forest patch size was tabulated for each data layer (1992 and 2001) within each of the 10,782 LTA polygons. Values were then obtained by subtracting 1992 patch size composition from 2001, to identify change in forest patch size composition. Thus, a positive result indicates an increased patch metric and a negative result represents a decrease. To facilitate interpretation, forest patch sizes also were grouped, into four groupings based on orders of magnitude. The Extensive Forest group has patch sizes equal to and greater than 1000 ha (1000 ha, 2500 ha, 5000 ha, and 10,000 ha). The Large Forest group included patch sizes equal to and greater than 100 ha (100 ha, 250 ha, and 500 ha). The Medium Forest group contains those patch sizes equal to and greater than 10 ha (10 ha, 25 ha, and 50 ha), and finally the Small Forest group contained all patch sizes smaller than 10 hectares (1 ha, 2 ha, and 5 ha). The forest patch grouping step was included to address this issue. Distribution of forest change was evaluated to locate areas of similar forest change intensity. LTA polygons exhibiting similar change intensity were evaluated.

Fragmentation was measured with a set of landscape metrics selected based on effectiveness in past studies and ease of interpretation (McGarigal & Marks 1995; Debinski & Holt 2000; Riitters et al 2000; Jones et al. 2001; Goodrich et al. 2002; R. Gardner, personal communication 2006). In contrast to many studies that employ landscape metrics (e.g., Riitters et al 2000; Jones et al. 2001; Li & Wu 2004), R. Gardner, personal communication (2006), stressed a strategic selection of metrics appropriate to goals of this study. The landscape metrics selected for this study, along with each paired fragmentation characteristic: (1) changes in edge, measured by change in edge density (ED), (2) changes in forest area, measured by the change in mean forest patch size (MPS), (3) changes in the forest edge to forest patch area ratio, as captured by a change in area weighted fractal dimension (AWFD), (4) change in the distance between forest patches or degree of isolation, as captured by the change in the mean nearest neighbor

(MNN), (5) change to core forest habitat, and (6) edge forest habitat, as captured by a changes to core vs. edge habitat composition (McGarigal & Marks 1995; Goodrich et al. 2002). Edge forest habitat is defined here as forested area within the first 100 m from a disturbed cover type (Temple 1984; Robbins et al. 1989; Debinski & Holt 2000; Goodrich et al. 2002) (Table 3.2).

3.2.3 Correlation Analysis

Pearson correlation was performed to explore relationships among the suite of fragmentation variables to identify redundancies. Using Minitab (2003) statistical software to assess variables, resulted in reduction of the initial 23 variables to 20. Variables that were highly correlated ($r > 0.850$) were eliminated from consideration in subsequent analyses (Williams 1979; Minitab 2003; Myers et al. 2006).

3.3 Results

3.3.1 Correlation Analysis

Pearson correlation analysis identified 20 landscape variables that were to be retained for analyses. Changes in total forest (TFOR), non-forest cover (NFOR) and core forest (CORE) were all correlated ($r > 0.850$) at $r > 0.999$ and the core forest metric was retained due to its interpretability and importance to other analyses. Edge forest (EDGE) and Edge Density (ED) also were correlated ($r > 0.870$) and edge forest was retained. Two other correlations were the small patch group (SMALL) with 5-ha patch size ($r > 0.723$) and medium patch group (MED) with the 50-ha patch size ($r > 0.784$). Both of these were above the $r > 0.700$ threshold suggested by Williams (1979), but below the $r > 0.850$ threshold applied by Myers et al. (2006).

3.3.2 Quintile Mapping and Evaluation

Changes in landscape metrics, between 1992 and 2001, were presented by a series of cumulative frequency maps using ArcGIS mapping functions (ESRI 2006). The Quintile function was used to map quintiles (five parts) by dividing the variable change data equally by LTA units from values calculated for each LTA polygon. After data exploration, quintiles were selected for ease of interpretability. Histograms of each metric were reviewed during data exploration and analysis. Generally, each variable exhibited a normal distribution and was centered near the zero “no change” value. Several variables had a disproportionate frequency near zero and in two instances the histograms were slightly skewed; mean forest patch size (MPS) toward the negative, indicating that more LTAs showed MPS getting smaller, and mean nearest neighbor (MNN) toward the positive, indicating that more LTAs contained forest patches that appeared to be moving apart. Mapping the cumulative frequencies of the landscape metrics proved effective for interpreting results. Each of the 10,782 LTA polygons were placed in one of five groups (+/- 2156 polygons) based on the cumulative frequency of change values (Table 3.3). Maps were prepared to highlight the LTAs exhibiting the greatest increases (top 20%) and greatest decreases (bottom 20%) in each metric change value. Gray was selected to identify positive trends on forest habitats (e.g., increases in forest patch size or decreases in forest edge) improving forested habitat condition. In contrast, I used black to map negative trends of forest habitats (e.g., decreases in forest patch size and increases in forest edge) identifying areas where fragmentation was increasing. White was used for the remaining areas that experienced less or zero change.

Maps revealed that four variables captured most apparent trends in forest change for Pennsylvania. These variables are core forest (CORE), edge forest (EDGE), mean forest patch size (MPS), and mean nearest neighbor (MNN). Core forest (CORE) change, as displayed in Figure 3.1, is changing in the northern half of Pennsylvania. Over half of the upper quintile of the LTA polygons showing increases in core forest (gray) composition exists in one contiguous group in north-central Pennsylvania. Adjacent and to the east was a large number of lower quintile polygons that indicated decreasing core forest (black) in that area. This area encompassed most of the Glaciated Low Plateau and separated along the western edge of the ecoregion. Some smaller areas increasing in core forest were seen in the Laurel Highlands of south-central Pennsylvania part of the Allegheny Mountain ecoregion, while areas losing core forest can be seen in southeastern Pennsylvania, north of Philadelphia, southwestern Pennsylvania, and in northwest Pennsylvania in High Plateau including the Allegheny National Forest. A contiguous area of LTA polygons that are decreasing (gray) in edge forest composition occur in the north-central part of the state, two additional, smaller areas occur in southeastern and southwestern Pennsylvania. Apparent increases in forest edge (black) appear in northeastern Pennsylvania in the Glaciated Low Plateau and Glaciated Pocono Plateau, and in the west-central Pennsylvania including the northern Pittsburgh Low Plateau and High Plateau (Fig. 3.2). Also, many valley areas of the Ridge and Valley ecoregion appear to be increasing in edge forest. Mean forest patch size (MPS) appears to be increasing (gray) in north-central Pennsylvania, Deep Valleys and eastern High Plateau ecoregions, and in the Allegheny Mountain (Fig. 3.3). Mean patch size is decreasing (black) in the valleys of the Ridge and Valley, most of the Glaciated Low Plateau, and in the High Plateau. Mean nearest neighbor values (MNN) are decreasing (gray) (i.e., forest patches are closer together) along a swath extending from north-central to south-central Pennsylvania in portions of the Deep Valleys, Allegheny Plateau, eastern Pittsburgh Low Plateau, and Allegheny Mountain (Fig 3.4). MNN is increasing (black) in western Glaciated Low Plateau, valleys of the Ridge and Valley, large areas of the Piedmont and, south-west Pittsburgh Low Plateau.

3.4 Discussion

3.4.1 Ecoregions and Watersheds

Ecoregion delineations are intended to provide spatial organization to ecosystem research and management (Omernik & Bailey 1997). Broad scale delineations are designed to encapsulate areas that share common structure and composition and that behave similarly to ecological processes. It follows that they would also respond similarly to regional perturbations. For land managers, ecosystem based planning can be organized by ecoregions with reliance that responses to within boundary treatments will be predicable (Bailey 1996). This, however, needs to be approached with caution. As Omernik and Bailey (1997) point out, ecoregion delineation has not been designed for regionalization of particular responses especially for finer scale phenomena.

Watersheds continue to be an important spatial structure for research and management when water quality and quantity issues are addressed (Omernik & Bailey 1997). As previously mentioned, the watershed concept is easier to define and more readily accepted. Countless studies have predicted natural and anthropogenic processes and informed as many management decisions. Along with their obvious effectiveness watersheds have problems as well. Topographic watersheds seldom capture the environmental features needed to predict quality and quantity without supplemental information (Omernik & Bailey 1997; Wardrop et al. 2005). Also, many areas, such as karst, glaciated or large flat areas are difficult to topographically delineate watersheds. Pennsylvania is glaciated in the north-west and north-east corners and limestone valleys in the Ridge and Valley ecoregion which complicate watershed work. Watershed delineations that rely on area based rules such as those mapped by the USGS hydrologic unit code (HUC) classification create inconsistencies when rules force watershed breaks at locations unassociated with natural pour points. These create "pass-through" watersheds that do not behave as true watersheds (Myers et al. 2006). Also, due to the terrestrial focus of this project, watersheds become problematic since forested habitats typically cross watershed divides and a watershed based assessment would separate what was otherwise contiguous habitat.

3.4.2 Landtype Associations for Ecological Study

Based on the above considerations, landtype associations became an important alternative for this spatially ecological assessment. Capturing much of the benefits of ecoregion mapping at a finer scale adds reliability to subsequent applications. The three major delineations described by Kong (2006) capture the ridge and watershed divide topographic features in the highland habitat (HH) and transitional terrace (TT) LTAs, and captures the hydrologic, side slope, and valley bottom features in the dual drainage LTAs. Areas of these polygons ranged from 39 to 2023 ha which is a scale well suited to detailed investigation. This study, however, did not exploit the potential detail embedded in the LTA delineation, but rather investigated spatial contiguity and regional patterns of the LTAs as classified by changes to forest composition and landscape metrics. One future challenge will be to discover relationships between individual LTAs and subtypes as characterized by forest change values to better understand the nature of the change. Kong (2006) included a comparison of the LTAs to the landscape vertebrate models completed by the Pennsylvania Gap Analysis Project (Myers et al. 2000). Results of this effort suggest that the addition of forest change data would be equally significant and would likely enhance these results.

3.4.3 Capturing Changes in Fragmentation Metrics

During development of landscape variables many studies involved evaluations of large groups of variables using various statistical approaches to determine which set best described the measured effects (Riitters et al. 1995; Opdam et al. 2003; Li & Wu 2004; Neel et al. 2004). To help develop this process some studies manipulated artificial, computer simulated, landscapes to help study metric behavior under controlled conditions (Gardner et al. 1987; Tischendorf 2001). For this study, I chose to reduce the number of landscape variables to focus on forest change (R. Gardner, personal communication 2006). Metrics were added to help capture habitat characteristics that forest change values could not address; such as edginess, perimeter vs. area relationships, and degree of isolation. Mean forest patch size was included as a convenient measure to capture change that would have missed by separate patch size classes. Grouping individual patch size changes into four groups (extensive, large, medium, and small) also helped interpretability. This was especially evident in the large and medium groups where patterns were difficult to visualize among the individual classes but emerged upon grouping patterns. This facilitates the application of these results to management scenarios where species with similar habitat area needs can be addressed by the groupings and, when necessary, the individual patch change classes can be used where more detail is required.

3.4.4 Management Scenarios

The Pennsylvania Gap Analysis Project (PAGAP) identified areas exhibiting high biodiversity potential and proposed management scenarios (Myers et al. 2000). PAGAP located areas of high vertebrate habitat coincidence based on vertebrate habitat models. Once modeling was complete, results were compared to existing stewardship lands (public and conservancy lands) to locate the “gaps” in the protection of biologically diverse areas. These new forest change data can enhance the PAGAP results by locating changing PAGAP areas. These areas of apparent change can be targeted for management activities. Pennsylvania has many species reliant on forest cover and many of these require interior forest for survival. For example, 100 ha has been reported to be an important area threshold for many avian species (Robbins et al. 1989; O’Connell et al. 1998; ELI 2003). By identifying 100 ha, and larger, forest areas in areas undergoing apparent change informed management for this habitat could be achieved (Fig. 3.5, 3.6).

Stewardship data can help to inform the forest change results. For example, by including the stewardship data with the core forest change map (Fig. 3.5), two patterns emerge. First, a significant amount of LTAs that are increasing their core forest in north-central Pennsylvania are within Bureau of Forestry land. This suggests that these areas under forest management are maturing and becoming less fragmented. Secondly, one reason why areas in north-east Pennsylvania appear to be losing core forest is that this region is nearly devoid of public lands.

Small forest patch classes seem to be especially problematic. As previously indicated (see Chapter 2) there is a strong likelihood that small forest patch class areas, <10 ha, have a >50% chance of being deforested. Single hectare patches have an 80% chance of being deforested. These results lead to review of the change in single hectare forest composition map (Fig. 3.7). The map shows many LTAs in the northeast, valleys of the Ridge and Valley including the Great Valley, most of the Piedmont, and the area around Pittsburgh increasing their composition of small, single hectare, forest patches. Based on Chapter 2 results, it follows that these areas are likely to be converted to non-forest cover and may have already been cleared. This change would be coupled with increased separation between forest patches (i.e., increase mean nearest neighbor) further complicating the between patch movements of wildlife.

Small forest patches are primarily in private ownership. Land ownership information coupled with information about land use behaviors of small forest woodlot owners will be necessary to better address the disappearance of small forest patches. A large proportion of the area of large forest patches is under public ownership and thus, the primary reason for these large forest patches is their history of forest management. The publicly owned forests of Pennsylvania are just a portion of that forest habitat management. A large percentage of the forested area of Pennsylvania exists in small forest patches (see Chapter 2). In many areas these forests provide habitat necessary to maintain forest habitat health in Pennsylvania.

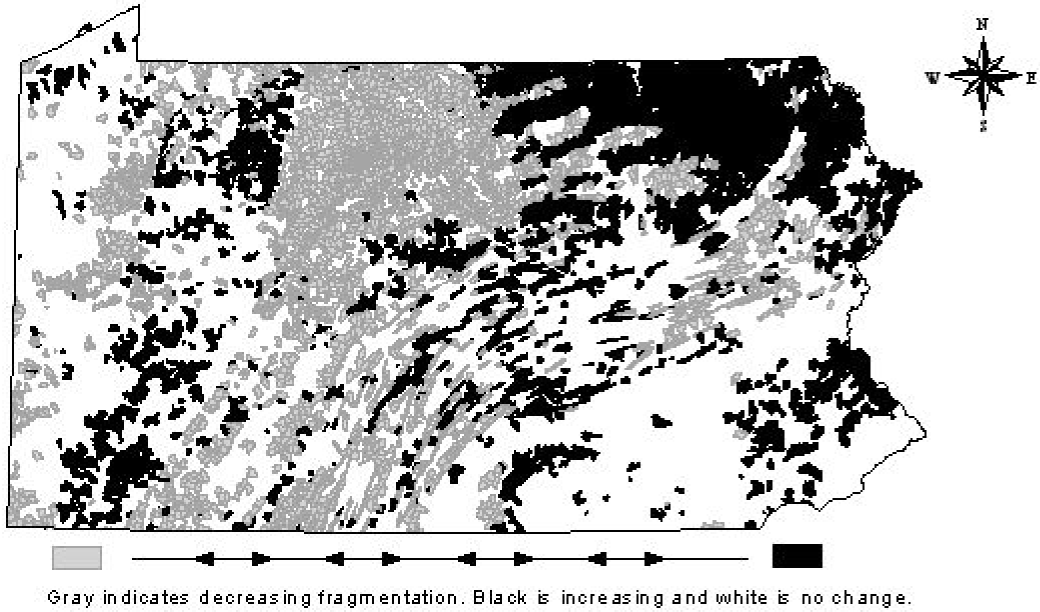


Figure 3.1 – Landtype Association polygons show increases in core forest (gray) in one contiguous group in north-central Pennsylvania. Areas that indicate decreasing core forest (black) are in northeast Pennsylvania in the Glaciated Low Plateau ecoregion. Other areas increasing in core forest can be seen in the Laurel Highlands of south-central Pennsylvania part of the Allegheny Mountain ecoregion, while areas losing core forest can be seen in southeastern Pennsylvania, north of Philadelphia, southwestern Pennsylvania, and in northwest Pennsylvania in High Plateau including the Allegheny National Forest. This quintile map shows the upper quintile, 20% of values, gray indicating a reduction in fragmentation and the lower quintile, 20%, black identifying increasing fragmentation. White areas are values in the middle, 60% of the values.

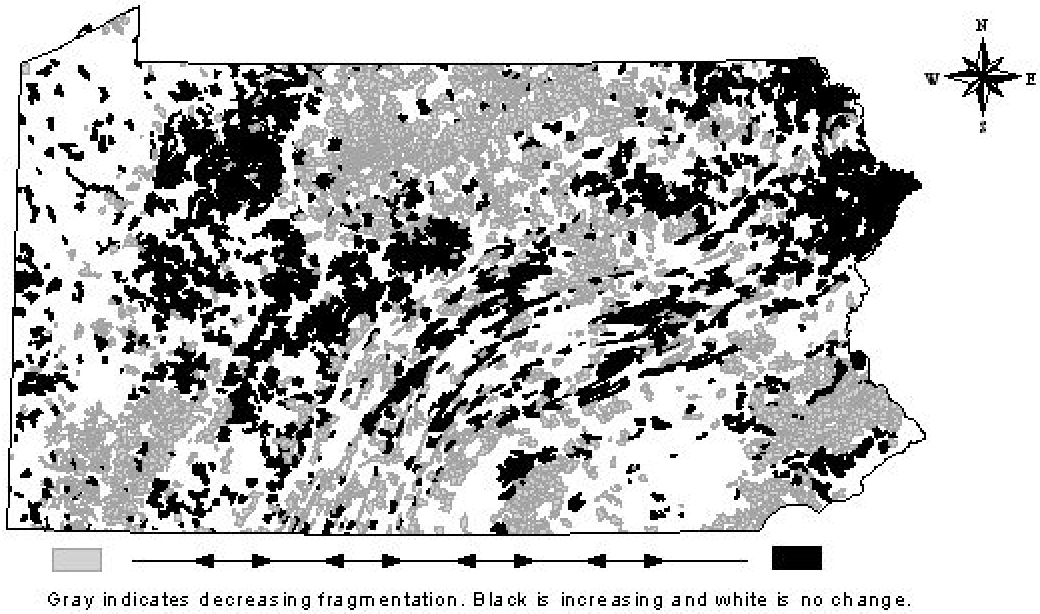


Figure 3.2 - LTA polygons that are decreasing (gray) in edge forest composition occur in the north-central part of the state, two additional, smaller, areas occur in southeastern and southwestern Pennsylvania. Increases in forest edge (black) appear in northeastern Pennsylvania in the Glaciated Low Plateau and Glaciated Pocono Plateau, and in west-central Pennsylvania including the northern Pittsburgh Low Plateau and High Plateau. Also, many valley areas of the Ridge and Valley ecoregion appear to be increasing in edge forest. This quintile map shows the upper quintile, 20% of values, gray indicating a reduction in fragmentation and the lower quintile, 20%, black identifying increasing fragmentation. White areas are values in the middle, 60% of the values.

Figure 3.3 – Mean forest patch size (MPS) increasing (gray) in north-central Pennsylvania, Deep Valleys and eastern High Plateau ecoregions, and in the Allegheny Mountain. MPS is decreasing (black) in the valleys of the Ridge and Valley, most of the Glaciated Low Plateau, and in the High Plateau. This quintile map shows the upper quintile, 20% of values, gray indicating a reduction in fragmentation and the lower quintile, 20%, black identifying increasing fragmentation. White areas are values in the middle, 60% of the values.

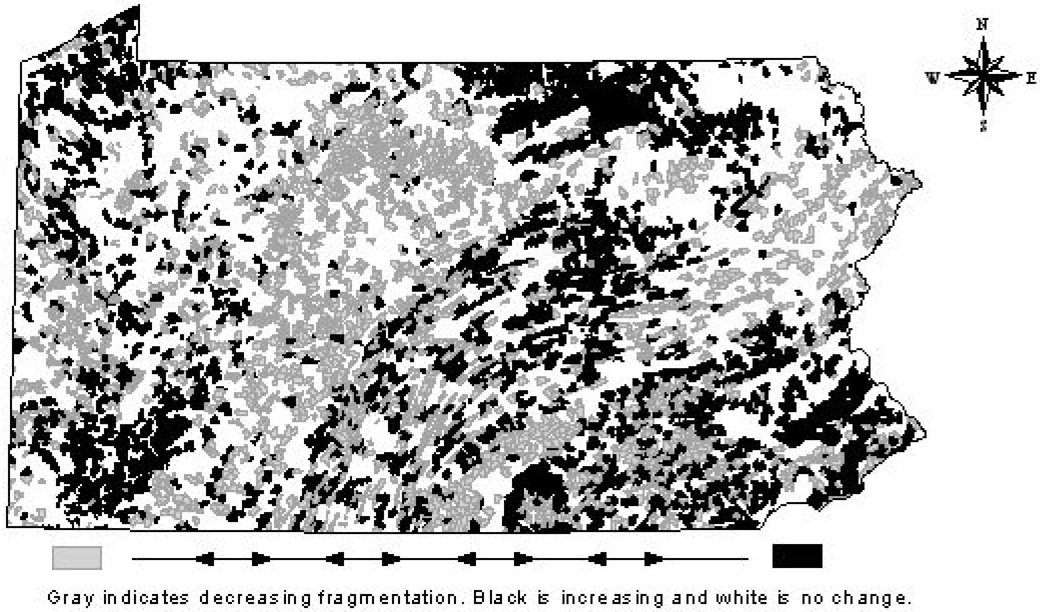


Figure 3.4 – Mean nearest neighbor (MNN) values decreasing (gray) (i.e., forest patches are closer together) along a swath extending from north-central to south-central Pennsylvania in portions of the Deep Valleys, Allegheny Plateau, eastern Pittsburgh Low Plateau, and Allegheny Mountain. MNN is increasing (black) in western Glaciated Low Plateau, valleys of the Ridge and Valley, large areas of the Piedmont and, southwest Pittsburgh Low Plateau. This quintile map shows the upper quintile, 20% of the values, gray indicating a reduction in fragmentation and the lower quintile, 20%, black identifying increasing fragmentation. White areas are values in the middle, 60% of the values.

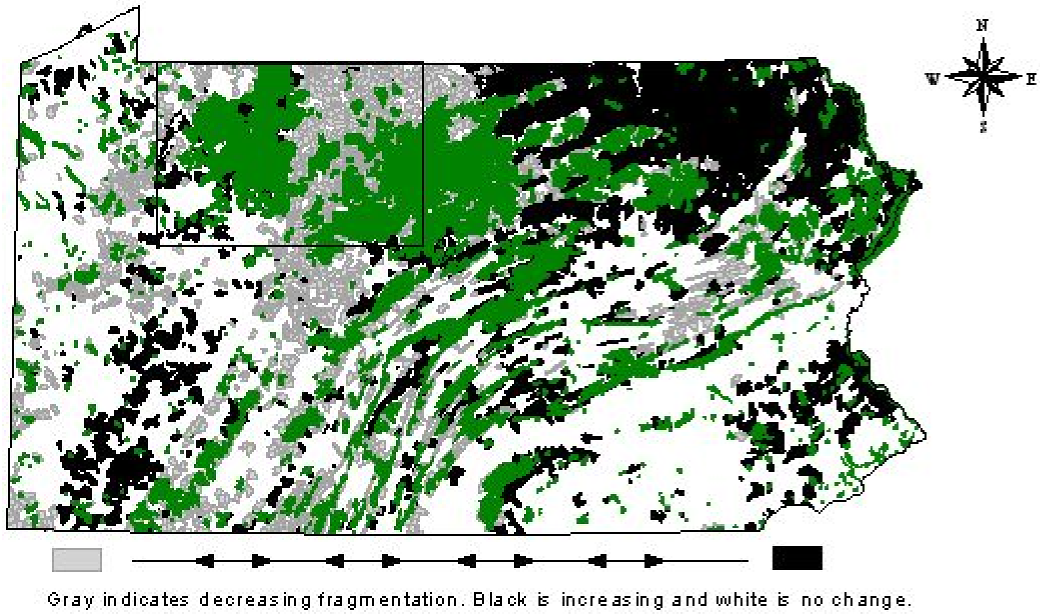


Figure 3.5 – Core forest is increasing (gray) and decreasing (black). To enhance interpretation, stewardship lands are included (green). This combination serves two purposes. First, it illustrates where to focus management activities and, secondly it helps to identify why certain conditions exist. Note that there is little managed land in northeastern Pennsylvania where core forest is disappearing. The box is the area of focus in Figure 3.6.

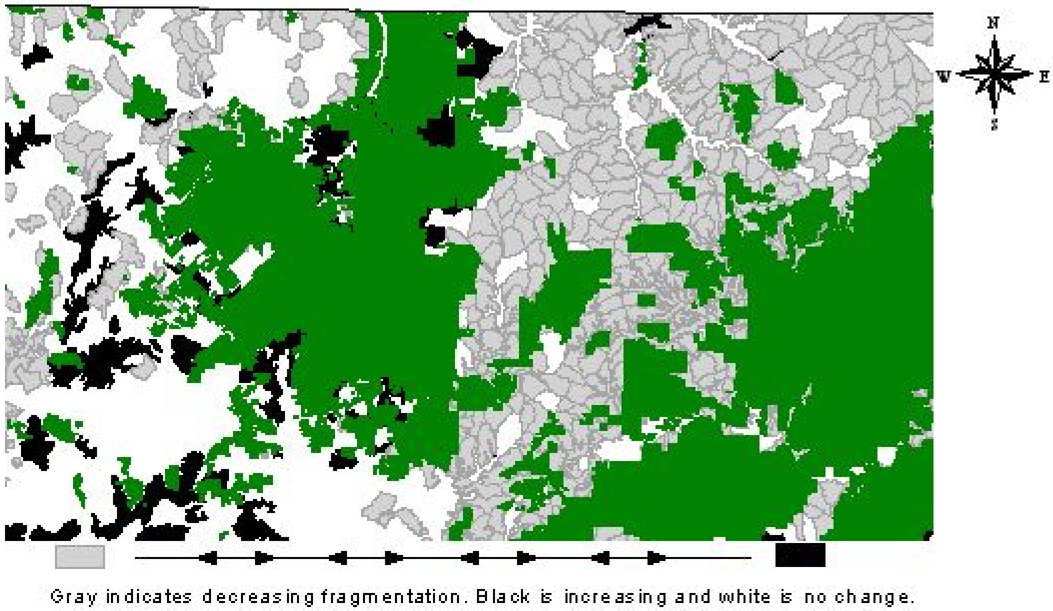


Figure 3.6 – Area is in north-central Pennsylvania including the Allegheny National Forest, to the west (left side of image), and the group of Pennsylvania Bureau of Forestry lands to the east. Displays how mapped forest change metrics could help agency officials set priorities for habitat management.

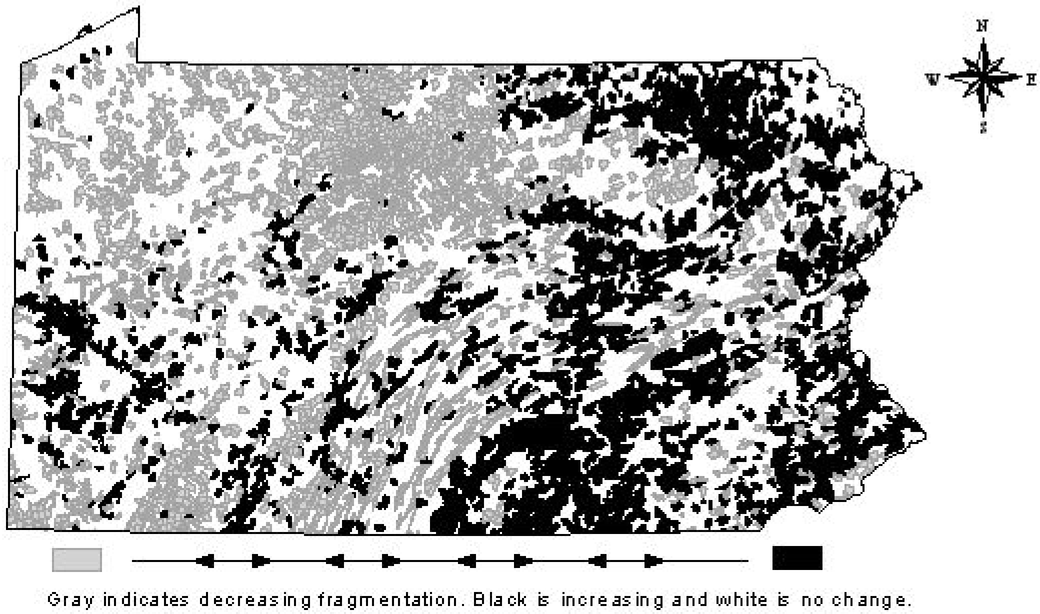


Figure 3.7 – Image maps the change to the single hectare forest patch class. The map shows many LTAs in the northeast, valleys of the Ridge and Valley including the Great Valley area, most of the Piedmont, and the area around Pittsburgh increasing their composition of small, single hectare, forest patches (black). The gray areas are decreasing their composition of 1 ha forest patches.

Table 3.1 – Subtypes defined by Kong (2006) to further classify the three landtype associations (LTAs) in Pennsylvania.

SUBTYPE	POLYGONS
Axial Aquaduct	255
Branching Basin	526
Convuluted Component	994
Elevated Exposure	365
Fluvial Facet	794
General Gradient	937
Hermit Height	175
Inclined Inflow	858
Local Lowland	264
Multi Mount	267
Original Outflow	691
Peripheral Plateaus	141
Regional Ridge	483
Side Step	215
Trough Terrain	147
Undulating Upland	2215
Veining Valley	1335
Water/Wetland	120

Table 3.2 – Summary descriptions of landscape metrics and forest patch size classes evaluated during this study.

Metric	Units	Description	Reference
Edge Density (ED)	meters/hectare	Sum of edge lengths, divided by total landscape area	McGarigal & Marks 1995
Mean Nearest Neighbor (MNN)	Meters	Sum of the distances to the nearest patch of the same type, divided by the number of patches of the same type (then convert to hectares).	McGarigal & Marks 1995
Mean Forest Patch Size (MPS)	Hectares	Sum of the areas (m2) of all patches, divided by the number of patches of the same type (then convert to hectares).	McGarigal & Marks 1995
Area Weighted Mean Fractal Dimension (AWFD)	1 LE (AWFD) LE 2 - Values approaching 1 represent simple landscapes and values approaching 2 represent complex landscapes	Sum of the corresponding patch type, of 2 times the log of patch perimeter (m) divided by the log of patch area (m2), multiplied by the patch area (m2) divided by total class area.	McGarigal & Marks 1995
Core Forest	Percent (%)	Composition (percent) of landscape that > 100 m into a forest patch from a disturbed edge.	Robbins et al. 1989; Debinski & Holt 2000; Goodrich et al. 2002
Edge Forest	Percent (%)	Composition (%) of landscape that is within the first 100 m into a forest patch from a disturbed edge.	Robbins et al. 1989; Debinski & Holt 2000; Goodrich et al. 2002
Patch Size Classes (ha) (1,2,5,10,25,50,100,250,500,1000,2500,5000,10,000, & 25,000 ha)	Class %. Represents minimum patch size < next patch size class (Except 1 ha class).	Composition (%) of each patch size class is reported.	ELI 2003 & Chapter 2

Table 3.3 – Quintile value limits for each forest patch size class and landscape metric evaluated in this study. Review of the quintile breaks reveals change trends in the class and metric values. Negative and positive shifts illustrate changes to metric values that indicate a loss within forest class area or a fragmentation change.

Patch Size Class	**Q1**	**Q2**	**Q3**	**Q4**	**Q5**
1 ha	-5.22--0.03	-0.03-0.24	0.24-0.7	0.7-1.34	1.34-8.49
2 ha	-10.5--0.38	-0.38-0	0-0.3	0.3-0.75	0.75-26.45
5 ha	-17.82--0.61	-0.61--0.01	-0.01-0	0-0.72	0.72-14.44
10 ha	-42.05--1.42	-1.42--0.01	-0.01-0	0-1.32	1.32-36.55
25 ha	-50.35--2.17	-2.17--0.01	-0.01-0	0-2.2	2.2-49.6
50 ha	-87.18--4.01	-4.01--0.01	-0.01-0	0-3.33	3.33-70.8
100 ha	-95.33--7.59	-7.59--0.61	-0.61--0.01	-0.01-0	0-97.2
250 ha	-99.64--3.13	-3.13--0.01	-0.01-0	0-4.59	4.59-99.83
500 ha	-100--0.07	-0.07--0.01	-0.01-0	0-4.74	4.74-100
1000 ha	-100--0.01	-0.01-0	0-2.27	2.27-13.15	13.15-100
2500 ha	-100--0.01	-0.01-0	0-3.27	3.27-19.84	19.84-100
5000 ha	-100--0.01	-0.01-0	0-2.73	2.73-21.15	21.15-100
10,000 ha	-100--0.01	-0.01-0	0-6.01	6.01-37.23	37.23-100
25,000 ha	-100--0.06	-0.06-0	0	0	0
Landscape Metrics					
ED (m/ha)	-103--10.3	-10.3--2.3	-2.3-2.9	2.9-10	10-163
MPS (ha)	-1527.6--16.1	-16.1--6.8	-6.8--3.2	-3.2-0.7	0.7-1731.4
AWFD	-1.13--0.02	-0.02-0	0-0.01	0.01-0.03	0.03-1.15
MNN (m)	-2373.1--2.98	-2.98-1.34	1.34-8.41	8.41-18.8	18.8-4890.8
Core	-68.23--10.79	-10.79--4.94	-4.94--1.31	-1.31-3.07	3.07-83.93
Edge	-46.21--6.56	-6.56--2.46	-2.46-0.22	0.22-3.78	3.78-44.17
Big Grp	-100--0.61	-0.61--0.01	-0.01-0	0-6.17	6.17-100
Large Grp	-98.84--12.41	-12.41--3.25	-3.25-0	0-4.92	4.92-100
Medium Grp	-67.17--5.48	-5.48--1.37	-1.37-0	0-3.1	3.1-70.8
Small Grp	-20.45--0.45	-0.45-0.05	0.05-0.77	0.77-1.97	1.97-42.72
Total Forest	-65.58--11.69	-11.69--5.43	-5.43- -1.35	-1.35-3.2	3.2-75.83

Chapter 4
Breeding Bird Response to Fragmentation Change in Pennsylvania

4.1 Introduction

4.1.1 Fragmentation Effects on Avian Habitat

Fragmentation and its resulting habitat loss are considered by many scientists to be the most significant threat to biodiversity today (Noss 1987a; Bennett 2003; Hilty et al. 2006). Habitat fragmentation has been described as the breakup and conversion of extensive habitats into smaller isolated habitat fragments too small to support their original species compositions (MacArthur & Wilson 1967; Wilcove 1987; Myer 1994; Fahrig 2003). Harris (1984) noted two components of fragmentation: (1) conversion of natural habitat in a landscape to other covers; and (2) separation and isolation of the remaining natural habitat into smaller patches. Forest fragmentation was identified as the most important threat to the health of natural resources of Pennsylvania (21st Century Report 1998; Goodrich et al. 2002)

Fragmentation of natural habitat has been shown to affect habitat use by birds (Temple 1984; Blake & Karr 1987; Robbins et al. 1989; Freemark & Collins 1992; O'Connell et al. 1998; Keller & Yahner 2007). Once habitat patch size is reduced beyond a species minimum area threshold, nesting and foraging success are affected and eventually some species no longer use reduced habitat patches that remain. Robbins et al. (1989) demonstrated that for many neotropical migrant birds both the likelihood of detection and relative abundance of the species increased as forest patch area increased. Blake and Karr (1987), in woodlots in Illinois, Freemark and Collins (1992), in Ontario, and O'Connell et al. (1998) and Keller and Yahner (2007), in Pennsylvania, all reported similar results. In addition to forest patch size, degree of isolation also was an important variable in these studies. Temple (1984), noted a similar patch size relationship for forest birds and reported that many forest birds were more likely to be detected when the area of core forest (forest area 100 m from an edge) was higher. In addition to the reported trends found as forest area decreased, edge effects were reported to increase, bringing habitat pressures brought on by generalist and parasitic species (Brittingham & Temple 1983; Yahner 1988; Yahner 1995; Rodewald & Yahner 2001).

4.1.2 Breeding Bird Survey Data for Ecological Response

The North American Breeding Bird Survey (BBS) is a volunteer-based "Citizen Science" effort that conducts bird censuses during the breeding season (May-July). Breeding Bird Survey (BBS) routes are 25 mile (40 km) long roadside transects and 3-min point counts are sampled at 0.5 mile (0.8 km) intervals (Robbins et al. 1986). Data are available through a website managed by the USGS Patuxent Wildlife Research Center (Sauer et al. 2005). Sauer et al. (2003) concluded that BBS data could be used to assess population trends from within Bird Conservation Regions, as described by Partners-in-Flight (Rich et al. 2004), similar in regional scale to Ecoregion Level-2 delineations (Bailey 1995; Omernik 1995). Data are subject to sampling bias from several sources. Sampling routes are located along roads and could disproportionately represent land covers. Common covers would be sampled more and restricted covers (e.g., wetlands, ridge-tops, etc.) might be omitted completely (Keller & Fuller 1995). Observer bias is also a problem and is particularly important when studying population trends (Sauer et al. 1994). Despite these problems, BBS data have been used effectively to assist ecological studies (Sauer et al. 2003; O'Connell et al. 2007). BBS data are organized and provided in several formats. Regional trends have been calculated and are provided from within both political and physical scales. Raw, unprocessed, data are available for each BBS route, 10-stop sub-routes, and, since 1997, individual stops (BBS 2007).

4.1.3 Response Guilds

Grouping of species into guilds that share similar behavioral and/or habitat needs has been conducted in past studies. Guilds are groups of species that exploit the same environmental resources, habitat, food, etc. in the same way (Simberloff & Dayan 1991). An advantage of guilds is that they address the difficulty of studying all species that live within an ecosystem and focus on specific groups with specific functional relationships (Simberloff & Dayan 1991). Functional, compositional, trophic, structural, and response guilds have all been suggested and tested in habitat inventory and management scenarios (Severinghaus 1981; Verner 1984; DeGraaf et al. 1985; Croonquist & Brooks 1991; O'Connell et al. 1998; O'Connell et al. 2000; Goodrich et al. 2002; Bishop & Myers 2005). Guild grouping is in contrast to taxonomic grouping and lends itself to judging a group response to change in the structuring factor that defines a particular guild. Wilson (1999) suggested two general categories of guilds (1) β (beta) guilds, those guilds comprised of species that will tend to occur in the same area or habitat, or have similar temporal cycles or distributions, and (2) α (alpha) guilds, guilds based on similar morphological or behavioral characteristics. Alpha guilds tend to have more within guild competition, thus, guild overlap is rare. Beta guild designation often involves same habitat space and members tend not to directly compete for resources (Wilson 1999).

Response guilds, a beta guild type, group the species by their predicted response to a habitat change or other environmental factors (O'Connell et al. 1998; Wilson 1999). Species in different response guilds will occur in different habitats and respond to environmental change differently. For example, if a forested area were to be reduced in size both individuals and species of a forest bird guild would be expected to be reduced proportionately, whereas a grassland or edge species guild would be expected to increase their numbers. One key aspect of response guilds is that once the impact of an environmental change is known for one of its members it can be inferred for the others (Severinghaus 1981; Simberloff & Dayan 1991). Past studies have suggested that comparisons of guild representation could be used to indicate the health of biological systems (Karr 1987; Brooks & Croonquist 1990; Canterbury et al. 2000). This has led to guilds being applied as environmental indicators (Verner 1984). An example of this is the bird community index (BCI) prescribed by O'Connell et al. (2000) and O'Connell et al. (2007).

Research Questions

- *What does forest change reveal about habitat condition based on BBS data change?*
- *Does change in guild species richness correlate with forest patch size change?*
- *Do changes in BBS data suggest habitat thresholds for the bird guilds?*

4.2 Methods

4.2.1 Response Guild Selection

For this study, four response guilds were created to assess change in habitat integrity for Pennsylvania. The guilds tested include: large forest obligate, grassland area sensitive, forest interior, and grassland habitat (Table 4.1) (O'Connell et al. 1998; Goodrich et al. 2002; Bishop & Myers 2005). Scale of resource use and movement/dispersal behaviors can be accounted for in response guilds. In this case, the response being tested is the response to changes in habitat area and habitat fragmentation. Guilds were selected that were expected to be most sensitive to fragmentation change. The landscape metrics (see Chapter 3) are well suited for testing habitat integrity as relating to habitat size and connectivity. I defined high guild presence for a guild by species presence for at least 50% of the guild members (Bishop & Myers 2005; O'Connell et al. 2007). To validate these analyses, areas with changing fragmentation were compared to breeding bird survey (BBS) trends recorded over the past nine years (see Appendix C).

4.2.2 BBS Data and Route Selection

Breeding Bird Survey data were obtained from the website of the Patuxent Wildlife Research Center (http://www.pwrc.usgs.gov/bbs/, BBS 2007) for all breeding bird routes in Pennsylvania. Each route was reviewed to determine if data were available for each time window coinciding with the two land cover data layers 1992-1994 and 1999-2001 (see Chapter 2). BBS routes with missing data were eliminated from consideration. Data were organized into10-stop summaries along each BBS route. Thus, five 10-stop summaries are available for each 50-stop survey route. Data are presented in tables containing species abundance within each 10-stop group along with abundance for the complete route. To account for species detection problems BBS data were pooled within each three-year period coinciding with the two erss of satellite imagery used in the land cover interpretations (O'Connell et al. 2007). BBS routes were chosen for testing from within regions of similar forest change intensity as identified by Chapter 3 results. Route placement was stratified, based on ecoregion, to account for the variety of physiographic settings found within Pennsylvania. This process resulted in 15 BBS routes being identified from within four physiographic settings over the variety of fragmentation patterns (Fig 4.1).

For each BBS route, one 10-stop segment randomly was selected for analysis. Guild presence data was tabulated for each of the 15 route segments and summarized in tables for each guild. Fragmentation metrics were collected from within the landtype association (LTA) polygon containing each BBS route segment. Fragmentation metrics, calculated in Chapter 3 for each LTA, were used as variables to test changes in guild presence that occurred between the two land cover eras, 1992 and 2001. Landtype Association polygons were selected that contained a majority of BBS stops of the 10-stop segment (Fig. 4.2).

4.2.3 Regression Analysis

Regression analysis was conducted to explore relationships between guild presence data and fragmentation metrics. Using Minitab statistical software (Minitab, State College, PA) the fragmentation metrics were first examined using a Best Subsets analysis. Best subsets was used (Neter et al. 1996; Minitab 2003) to help identify the set of variables most likely to connect fragmentation metrics with changes in guild presence. Best Subsets is designed to select the set of variables that best explains variation in a response variable. Minitab (2003) computes statistics for each quantity of variables. The best model with one variable is the variable that has the highest R^2 value. After the best one-variable model is computed then two variables combinations are selected. Then, the three-variable model with the highest R^2 is calculated and so on.

The R^2 value is not the only statistic used for selecting the best set of variables. The R^2 value always increases with the addition of each new variable to the model. To select the best set of variables the C_p value must be evaluated as well. The C_p value is a measure of variance. A low C_p indicates that a model is relatively precise. A good C_p is a value that is close, or equal, to the number of variables currently being tested. Once a low C_p value is realized, within a best subset computation, the C_p will begin to get larger with the addition of each new variable. Therefore, the best subset model is the model with a high R^2 value paired with a low C_p value. The resulting set of variables for each guild was processed in a linear regression model to identify the amount of change to guild abundance that might be attributed to the variation among fragmentation metrics.

4.3 Results

4.3.1 Determining Guild Presence

Guild tallies were conducted by constructing a series of database tables that, temporarily, could be linked and unlinked depending on the desired tally. These tables contain >1000 records with each necessitating an automated approach. Separate tables for each guild were created containing records for all active BBS routes in Pennsylvania. A separate route table was constructed for each guild that provided a way to link the individual selected routes to the larger guild tables to individually select breeding records for each route within each time period. These were tallied systematically to determine total guild presence values. Totals were based on a simple yes/no presence of a member within the time windows. Guild abundance was not included for this study.

Recognizing that, based on potential ranges, not all members of any guilds occur throughout Pennsylvania (Brauning 1992), an adjustment was made to evaluate percent guild presence. Data were tabulated for every active BBS route in Pennsylvania. Tallies were run for the four guilds to determine the BBS route with the maximum number of guild members present. That number was then used to calculate percent presence for individual routes (Table 4.1).

Once the guild presence tallies were complete for each time period, 1991-1994 and 1999-2001, and the adjustment value was known, guild presence was calculated for each 10-stop segment sampled for each BBS route. Large forest obligate and forest interior guilds both had one route above 50% for the 1991-1993 period, and that total increased to four routes above 50% for 1999-2001. Grassland area sensitive had three routes above 50% for the first period, but dropped to one route for the second period. Grassland habitat had five routes above 50% in 1991-1993 and dropped to three routes in 1999-2001 (Table 4.2a, 4.2b, and Appendix D). These percent presence values for each time were then differenced to determine the change that occurred between the sampling windows. The change values were then used as response variables for regression analysis.

4.3.2 Regression Analysis

The results of the best subset analyses processed with Minitab statistical software (Minitab 2003) are listed in Table 4.3. Each of the four response guilds were tested against nine independent variables to establish which subset of variables would explain the greatest amount of variation for each response guild. Minitab (2003) selected the two models from each quantity of variables that had the highest R^2 value. From the 17 potential models, the model with the paired high R^2 value and low C_p value was selected for further analyses. Among the four guilds, the mean forest patch size (MPS), core forest (CORE), and the large grande forest group (GRANDE) were included in three of the four linear regression models. Area weighted fractal dimension (AWFD), medium forest group (MED), and smallest forest group (PEQ) were all in two models and edge forest (EDGE) and the largest vente forest group (VENTE) were each in one model. The mean nearest neighbor (MNN) variable was not selected. One linear regression model, grassland area sensitive (GAS) birds, showed significant results (R^2=69.1% and P=0.013). The other models, grassland habitat (R^2=44.0% and P=0.176), large forest obligate (R^2=53.3% and P=0.160), and forest interior (R^2=36.0% and P=0.626), were not significant (Table 4.4).

A scatter plot graph was plotted for each guild change value against the variable selected as the best single predictor from the best subset analyses. The medium forest patch size group (MED) was identified as the single predictor for both forest interior and grassland area sensitive guilds, large forest patch size group (GRANDE) and small forest patch size group (PEQ) were picked for large forest obligate and grassland habitat guilds, respectively (Table 4.4). These two variables, PEQ and GRANDE, consistently accounted for more variability in the response variables. The forest interior guild graph shows guild presence trends decreasing as composition of medium forest patch size group (10 ha, 25 ha, and 50 ha) increased (Fig. 4.3a). The large forest obligate guild graph shows guild presence increasing as the large grande forest patch size group (100 ha, 250 ha, and 500 ha) composition increases (Fig. 4.3a). The grassland area sensitive guild graph shows the guild presence decreasing as the composition of the medium forest patch size group (10 ha, 25 ha, and 50 ha) increases (Fig. 4.3b). The grassland habitat guild graph shows that guild presence decreases as the smallest forest patch size group (1 ha, 2 ha, and 5 ha) composition increases (Fig. 4.3b).

4.3.3 Guild Presence and Regional Patterns for BBS Route Locations

Breeding Bird Survey routes 72006, 72017, and 72014 are all located in north-central Pennsylvania in areas showing apparent decreases in regional fragmentation. Based on an initial review of data (Chapters 2 and 3), it would be expected that these routes would show stable to increased responses in the forest guilds and stable to decreased change responses in the grassland guilds. The combination of change to guild presence values, and initial guild member totals both indicate a decrease in fragmentation. Guild presence values for large forest obligates and forest interior guilds both show increases, with the exception of 72017 for large forest obligates, indicating that each guild member count increased for these routes during the nine year time span. The change values are (FINT-CH = 12.9, 9.7, and 0.0 respectively) for forest interior birds and (FOB-CH = 11.7, 17.6 and -17.6) for large forest obligates. At the same time, both grassland guilds recorded low guild presence counts and indicated no change between dates (Figs. 4.2a, 4.2b).

Breeding Bird Survey routes 72009 and 72058 are all located in west-central Pennsylvania in areas showing varying amounts of regional fragmentation. Based on an initial review of data from Chapters 2 and 3, it would be expected that these areas would exhibit varying responses in both guild types, forest and grassland guilds. The combination of change in guild presence values, and initial guild member totals both indicate a region of active change. Guild presence values for large forest obligates and forest interior guilds increased for 72009 (FOB-CH=23.5, FINT-CH= 32.3) and decreased for 72058 (FOB-CH= -23.5, FINT-CH= -22.5) where grassland guild counts were both initially and, more recently, high and stable. One exception to note is that the grassland habitat guild decreased for 72058 (GRASL-CH = -18.2) (Figs. 4.2a and 4.2b).

BBS routes 72097 and 720181 are located in south-western Pennsylvania in the areas impacted by the suburban sprawl of Philadelphia. Based on regional trends, it would be expected that the forest guilds and the grassland area sensitive guild would all be decreasing in guild presence and the grassland habitat guild would be relatively stable. It is interesting to note that this is not the case. Both routes exhibited stable to slightly increasing grassland area sensitive bird totals, decreasing change totals for both grassland habitat and forest interior guilds and slight increases to the large forest obligate birds. Results are (GRASL = -18.2, FINT = -3.22 and FOB = 5.9) for 72097 and (GRASL = -27.3, FINT = -3.23 and FOB = 5.9) for 72181 (Figs.42a and 42b).

BBS routes 72031, 72038, and 72901 are located in the glaciated part of north-east Pennsylvania that, based on initial review of data form Chapters 2 and 3, has experienced regional decreases in forest area coupled with increases in fragmentation. It would be expected that these areas would show changing responses in both guild types, forest and grassland guilds especially showing negative change for large forest obligates. With the exception of a decrease in the change value for forest interior birds, forest guild counts and change values for all three routes are stable and increasing, and the grassland birds were initially high and exhibit stable to decreasing values. Guild change results are (GAS = -14.3, GRASL = -9.09, FINT = -3.23 and FOB = 0.0) for 72031, (GAS = 0.0, GRASL = 0.0, FINT = 3.23 and FOB = 5.9) for 72038, and (GAS = 0.0, GRASL = -9.09, FINT = 12.9 and FOB = 11.8) for 72901 (Figs. 4.2a and 4.2b).

BBS route 72036 is located in a valley area of the Ridge and Valley and, based on initial review of data from Chapters 2 and 3, would have been experiencing a history of fragmentation. Guild counts and change values point to this pattern with high and stable counts for grassland guilds and low and decreasing counts for forest guilds. Results are (GAS = 14.3, GRASL = 0.0, FINT = -3.23 and FOB = 5.9) (Figs. 4.2a and 4.2b).

4.4 Discussion

4.4.1 Breeding Bird Survey Response

This study was designed to explore ecological relevance between changes in fragmentation metrics and changes in breeding bird survey (BBS) data over time across expansive landscapes. Although results for three of the four guilds were statistically not significant, trends suggest that there is a relationship between the forest change and BBS change. Deliberate decisions were made to target the study to answer one question. Do the calculated landscape level fragmentation metrics predict changes recorded in the BBS data? As mentioned before, BBS data are subject to sampling bias from several sources. For this study, BBS data were pooled around dates matching the two satellite-based land cover data sets. This was done to alleviate some of the issues of “observer bias” as discussed by Sauer et al. (1994). Bird detections were pooled across three years and recorded as a positive response for that species. The 15 test routes were selected based on mapped fragmentation trends.

The attempt was to find routes in areas undergoing change and to cover the variety of those changes. The response guilds were chosen so that their collective membership would be sensitive to detectable changes in landscape-level fragmentation metrics. This alleviates the sampling bias discussed by Keller and Fuller (1995) by focusing on the common land covers (i.e., forest change). If, for example, a wetland guild were to be tested, detectable changes would not be expected. Finally, selecting response variables that had been calculated within the landtype association (LTA) boundaries alleviates the concern that many landscape metrics (McGarigal & Marks 1995) behave unpredictably when calculated in defined landscapes too small relative to the grain of land cover data. O’Connell et al. (2002, 2007) tabulated land cover composition for 500-m radius circles around BBS points to assess current condition. Additional landscape metrics were not included. Following similar logic, the same metrics as calculated in Chapter 3 for the LTA boundaries could be processed for the

500-m radius areas around the selected BBS routes. However, based on results from Riitters et al. (1995), and Li and Wu (2004) this would lead to inconclusive results. Their results indicate that metrics that report area-perimeter and distance relationships are irregular when defined landscapes are small relative to data grain. In Chapter 3 three of the metrics that display apparent regional patterns, edge density, area weighted fractal dimension, and mean nearest neighbor, are in one of these groups. Rather than use the 500-m radius circle, the LTA boundaries provide a convenient alternative. They define a region similar in scale to the BBS data while reducing the likelihood of these reported data scale problems in the fragmentation metrics.

4.4.2 Land Cover Data and BBS for Regional Assessment

The nationally structured biodiversity project Gap Analysis relies heavily on 30-m scale land cover data as the basis for habitat modeling (Scott et al. 1993; Myers et al. 2000). The goal of the modeling effort is to predict vertebrate habitats and locate those areas with the highest diversity for future management. The National Land Cover Data (NLCD) has been a source for these data and recently has released a new national land cover data layer (Homer et al. 2004). As this new land cover data is evaluated the ability to study change between the dates becomes possible. Within delineated Bird Conservation Regions (BCRs), Partners-in-Flight has identified target species to guide conservation efforts (Sauer 2003; Rich et al. 2004). Studies, like this, could help identify areas of change to estimate threat and provide focus to these efforts.

Breeding Bird Survey data provide the only nationally-based annually-surveyed biological inventory. Sauer et al. (2003) concluded that BBS data could be used to assess population trends from within BCRs. O'Connell et al. (1998) used species occurrence in forest patches varying in size to help establish minimum forest area thresholds to help create guilds. Using BBS data, similar methods could be applied to forest change areas to further refine guilds to explore sensitivities to change. Past investigations have shown that certain bird species respond to different habitat thresholds, but what are the effects of the rate of fragmentation change on avian species?

4.4.3 Regional Evaluation of Guilds

The BBS routes selected for this study were chosen based on a combination of regional location, loosely stratified by ecoregion and local forest fragmentation trends as mapped via classified LTA boundaries. Although, guild presence change was not statistically significant for three of the four guilds, this could be due to many reasons. One possible explanation is that the lack of response could be due to the small sample size. Fifteen routes is small especially when stratified across Pennsylvania.

Interpreting these results is challenging and three important issues need to be considered. First, a "no change" response in guild presence may be exactly what would be expected to occur. If a route is located in an area that has been consistently in one land cover for the nine years, then it is not likely to register a change in guild presence. In some cases, no change in the forest birds is a positive response indicating that the actual counts are more important to evaluate than the change between dates such that no ecologically measurable change in fragmentation occurred. Secondly, guilds selected were done so to target forest change and fragmentation change, both positive and negative. This ignores some of the generalist, less habitat specific, species that are likely to be increasing in a developing area. This would be especially true if a route was in or near an urban area. Last, forest change metrics do not change independently and need to be evaluated as a group not individually. Using one or two metrics does not reveal the whole story. In some cases, for example, an increase in core forest area matches with a decrease in edge forest, indicating that forests are regenerating in areas and edges are merging together. In other regions an increase in core forest may be offset by a decrease in edge where edge decrease indicates complete loss of smaller woodlots.

A finer scale study would be needed to verify these effects. Based on this study, areas that are demonstrating this pattern are those that have been under continued suburbanizing pressures, especially those in southeastern Pennsylvania. Some woodlots have been allowed to expand, increasing core forest, while other woodlots have been removed to provide space for new housing. Review of more detailed imagery would be valuable to complete this study.

4.4.4 Treatment of Exceptions

Results suggested that referring to all of the data sets created in this study, including those from Chapters 2 and 3, would be beneficial to interpreting guild responses. This includes accounting for both sets of guild count values. Three BBS routes were deliberately omitted from the results section to serve as examples to illustrate the need to completely examine all data to understand the observed guild change responses. The BBS route 72189 is located in south-central Pennsylvania in the Laurel Highland area. By solely interpreting the guild change response (GAS = 0.0, GRASL = 18.18, FINT = 0.0, and FOB = 5.9) it could be concluded that this is a grassland area (note increase in grassland habitat guild) with no changes in the level of fragmentation. Closer review, based on guilds count values, reveals a stable forested area, indicated by the highest (both pre- and post-) count values from both forest guilds and that it also seems to have some stable grassland areas as evidenced by the stable counts from the grassland area sensitive guild. Upon review of the fragmentation metrics, there is further support for a consistently forested condition as evidenced by the increases in mean forest patch size, core forest, and the largest VENTE forest, matched by decreases in the two small forest groups (MED and PEQ) and mean nearest neighbor all indicators of a decreasing fragmentation trend. This trend is slightly offset by a decrease in the large forest group GRANDE and an increase in the edge composition that could indicate increased fragmentation but appears to be out weighed by other metrics. The piece to consider looks at local land use. Upon review of the public land information, it was discovered that most of the route segment is within Ohiopyle State Park and thus, is managed for forested habitats.

The BBS route 72905 is located in central Pennsylvania in the Ridge and Valley ecoregion. By solely interpreting the guild change response (GAS = 0.0, GRASL = 0.0, FINT = 6.45, and FOB = -5.9) is could be interpreted that this route was located in a stable area that may be losing some forested area, as evidenced by the slight decrease in the change value for the large forest obligate guild. This is supported further by the pre- and post- bird counts of zero for both grassland guilds. The landscape metrics support the forest condition as well, with increasing values for mean forest patch size, core forest, and VENTE forest group, and decreasing values for mean nearest neighbor, edge forest, and the two small forest groups all indicating decreasing fragmentation. The only inconsistent value is the small decrease in the large forest obligate guild. This, perhaps, can be accounted for by examining the guild members with respect to the actual BBS 10-stop segment tested. In this case the 10-stop segment runs along the top of Tussey Ridge, south-east of State College, PA (Centre County). Guild members in the large forest obligate guild while generally having affinity for big forests are reported to avoid ridge-tops for nesting (S. Hoffman, personal communication 2004). This could account for the ambiguous value.

The last route for closer review is BBS route 72080 located in the Great Valley of Pennsylvania (Ridge and Valley ecoregion) northeast of Harrisburg, PA. General knowledge of the region and review of guild change values (GAS = -28.6, GRASL = -18.2, FINT = 3.23, and FOB = 5.9) would indicate a historically grassland/agricultural area with small amounts of forest regeneration as indicated by slight increases in the forest guilds. Review of the pre- and post- bird counts reveals that forest guild change is a result of the counts increasing from only zero to one bird for both guilds. This further supports the land use history of the area. The fragmentation metrics are inconclusive as well with little changes reported except for one metric. Of all the routes in this study, route 72080 has the highest increase to the nearest neighbor value indicating increasing fragmentation. Review of the 2001 land cover layer adds the final evidence. The 10-stop segment is located in a suburban area which accounts for the reported decrease in both grassland bird guilds. As mentioned before, this points to guild membership and, in this case, the guild selected for this study. As previously mentioned, the four guilds evaluated for this study, were selected to test avian response to fragmentation change. In this case the area is already fragmented and the decreases recorded in the grassland guilds are probably in response to the grassland becoming urbanized. A generalist guild with members tolerant of urban conditions would likely yield positive results. Thus, there may be land use and landscape configurations at the local scale that override the regional fragmentation patterns observed. Understanding the scales at which data are collected, analyzed, and interpreted is necessary before confirming the results.

4.4.5 Landtype Associations and the Breeding Bird Survey

Ecoregion mapping is designed to encapsulate areas that share common climatic and vegetation characteristics and then be used to guide research and inform management in regions of similar ecological condition (Bailey 1995; Omernik 1995). Landtype associations (LTA) are hierarchically nested within ecoregions and delineated at a finer scale to capture local ecological variation (Myers 2000; Kong 2006). For my study, LTAs serve as a boundary to assess forest fragmentation change. Although not tested in this study, the assumption is that the ecological processes influencing forest cover within each LTA are complementary. Thus, forests within the same LTA would respond predictably to ecological changes and perturbations. For this study the LTAs served as a means to capture fragmentation change based on forest cover recorded for two time periods separated by nine years. Review of mapped fragmentation metrics within LTAs proved effective for displaying regional fragmentation trends evident for most of the calculated metrics.

Sauer et al. (2003) concluded that BBS data could be used to assess population trends from within the Bird Conservation Regions described in Partners-in-Flight guidelines (Rich et al. 2004). As previously mentioned, BCRs are similar in regional scale to ecoregion delineations (Bailey 1995; Omernik 1995). Thus, trends in fragmentation metrics recorded for the LTAs would have direct relevance to initiatives that are based on BCR boundaries. Several of the regional, and Pennsylvania level, PIF recommendations involve forest habitat management and fostering programs to curtail fragmentation. Combining the fragmentation trends visible among the LTA boundaries with occurrence data from the BBS helps to identify known locations in need of management.

For this study, BBS routes were selected to test avian response to fragmentation change using a response guild approach. Once BBS data availability for both land cover eras was determined, test routes were selected in specific locations to exploit fragmentation trends as mapped by the LTA boundaries. Fifteen routes were located to capture decreasing, increasing, and irregular fragmentation stratified across the four, primary, ecoregions of Pennsylvania; the Piedmont, Ridge and Valley, and Glaciated and Non-glaciated Allegheny Plateau. Several methods were used to alleviate known biases in the BBS data. Observer bias and detection problems discussed by Sauer et al. (1994) were addressed by pooling data among the 3-year datasets (O'Connell et al. 2007). The selection of 10-stop segments helped smooth the landscape heterogeneity that exists along a complete 25-mile route and provides responses focused at a scale closer to that of the LTA boundaries. Finally, employing a response guild approach alleviates the need of evaluating the presence or absence of a species and allows the proportional presence of guild members to establish the integrity of habitat for that guild.

4.4.6 Future Work

Past studies have suggested that comparisons of guild representation could be used to indicate the health of biological systems (Karr 1987; Brooks & Croonquist 1990), which has led to application of guilds as environmental indicators (Verner 1984; O'Connell et al. 2000; O'Connell et al. 2007). An example of this is the Bird Community Index (BCI) prescribed by O'Connell et al. (2000). One key aspect of response guild usage is that once the impact of an environmental change is known for one of its members it can be inferred for the others (Severinghaus 1981; Simberloff & Dayan 1991). As suggested by Severinghaus (1981) and Simberloff and Dayan (1991), guild counts record not only which specific members are present, but help identify quality habitat for the entire guild.

Although not a strict application of guild use as a biological indicator, my study helps to further develop that process. Up to now fragmentation change metrics have not been applied over large areas (i.e., the state of Pennsylvania) at specific locations (i.e., 10-stop BBS segment scale). Regional trends have been compared to regional land cover from static points in time (Sauer et al. 2003). When ecological indicators, such as the BCI, have included land cover data, it was also from one point in time. Including fragmentation change metrics between two dates helps to identify areas in transition and coupling that change with guild counts helps to target that effect on habitat quality.

This suggests that adding a fragmentation change value is a better way to understand rate of change as an impact of ecological systems. Potentially adding a change metric to the BCI or creating a separate index of change or change vector would be useful to enhance this research. One line of research that this type of variable would provide would be a better understanding of the rate at which fragmentation change affects breeding success. For this study I tried to match the BBS data to the land cover dates and then tested BBS response. Perhaps avian response to land cover change is delayed, thus by shifting to later dates to summarize BBS data a lag period could be accounted for and the delayed response addressed in the analyses.

Complete processing of the BBS routes for Pennsylvania would provide a more complete assessment of avian response for Pennsylvania. For this study I specifically selected one 10-stop segment in each of 15 BBS routes located in areas that displayed regional fragmentation patterns. By analyzing such a low sample set (N=15) is a likely reason why my results were not statistically significant. There are approximately 25 additional routes that had sufficient BBS data to test and by including them a more complete view of BBS response to fragmentation change could be evaluated.

A potential alternative method to tracking changes in total guild members would be one that uses methods suggested in O'Connell et al. (2000) and O'Connell et al. (2007) to calculate the bird community index (BCI) scores for the BBS routes. Indices could be calculated for each land cover era and differenced to track change. The BCI was developed using a guild approach that assigns a score based on quality of habitat needs of each species. The key difference is that the BCI assigns values for all birds present, not just birds from a specific guild. Thus, even if there is no change within the membership of a specific guild, if the avian community has changed then it will be reflected in the final BCI score.

As discussed in many studies, continuing and improving satellite-based land cover interpretation is critical for habitat management worldwide (Vogelmann et al 1998; Scott et al. 1993; Stoms & Estes 1993; Myers et al. 2000; Homer et al. 2004; O'Connell et al. 2007). In September 2007 more recent land cover data became available for Pennsylvania (Warner et al. 2007). The ability to add a third set of forest fragmentation metrics to continue tracking change is now possible. The question of testing the possible avian response lag would be assisted by these new data. Also, including a third, similarly classified, land cover data layer provides an extended period to test for effects of forest fragmentation.

The Landsat-based land cover used for this work has a 30 m x 30 m pixel size. It is not possible to know what is not detected inside these 900 m^2 pixels without using imagery that has been collected at finer resolutions. Calculating fragmentation metrics for the state-wide extent was computationally difficult and individual analyses normally took days to, sometimes, weeks to complete. Thus, to complete a state-wide analysis using a finer resolution image source, such as SPOT data, at 10 m^2 , or PAMAP imagery, at 1 m^2 , while certainly valuable, becomes computationally challenging.

In any investigation the scale of the base data becomes integral to the detection of land cover change. As the resolution becomes finer smaller objects become visible. At the 30 m x 30 m pixel the best that can be detected is cover that occupies the majority of area of the pixel. Thus, if most of the ground is covered by deciduous trees then that is the class even if other covers are present. Using 1 m x 1 m pixel area images individual tree canopies become visible and the challenge becomes selecting the best interpretation rules. The radiometric classification would have several pixels to cover a single tree canopy. This adds challenges to data management when classifying large areas is being considered (G. Baumer, personal communication 2007).

Table 4.1 – Response guild name, abbreviated guild name, total members. The Adj_No. column represents the highest guild member count for any BBS route in Pennsylvania.

Guild Code	Guild Name	No. of Members	Adj_No.
FOR_INT	Forest Interior	46	31
GRASL	Grassland Habitat	21	11
FOR_OB	Large Forest Obligate	23	17
GAS	Grassland Area Sensitive	11	7

Complete reference list for guild assignments

Bishop & Myers (2005) - Associations between avian functional guild response…..
Goodrich et al. (2002) - Wildlife habitat in Pennsylvania: past, present, and future
O'Connell et al. (1998) – A community index of biotic integrity for the Mid-Atlantic highlands.
Brauning (1992) – Atlas of Breeding Birds in Pennsylvania.

Table 4.2a – Breeding Bird Survey data for response guilds grassland habitat birds and grassland area sensitive birds for each survey route. The change column reports the percent change between the two dates.

Grassland Habitat Guild – 21 Members

ROTNO	STOPS	GP_9193	GP_9901	Adj_No	PGP_92	PGP_01	Change
72038	1-10	4	4	11	36.36	36.36	0.00
72031	41-50	4	3	11	36.36	27.27	-9.09
72901	11-20	5	4	11	45.45	36.36	-9.09
72014	41-50	0	0	11	0.00	0.00	0.00
72006	41-50	3	2	11	27.27	18.18	-9.09
72017	31-40	0	0	11	0.00	0.00	0.00
72036	41-50	7	7	11	63.64	63.64	0.00
72181	31-40	4	1	11	36.36	9.09	-27.27
72009	41-50	4	6	11	36.36	54.55	18.19
72080	11-20	6	4	11	54.55	36.36	-18.19
72905	41-50	0	0	11	0.00	0.00	0.00
72058	1-10	6	4	11	54.55	36.36	-18.19
72097	1-10	6	4	11	54.55	36.36	-18.19
72048	41-50	8	5	11	72.73	45.45	-27.28
72189	21-30	2	4	11	18.18	36.36	18.18

Grassland Area Sensitive Guild – 11 Members

ROTNO	STOPS	GP_9193	GP_9901	Adj_No	PGP_92	PGP_01	Change
72038	1-10	2	2	7	28.57	28.57	0.00
72031	41-50	3	2	7	42.86	28.57	-14.29
72901	11-20	3	3	7	42.86	42.86	0.00
72014	41-50	0	0	7	0.00	0.00	0.00
72006	41-50	2	2	7	28.57	28.57	0.00
72017	31-40	0	0	7	0.00	0.00	0.00
72036	41-50	4	5	7	57.14	71.43	14.29
72181	31-40	1	2	7	14.29	28.57	14.28
72009	41-50	4	4	7	57.14	57.14	0.00
72080	11-20	4	2	7	57.14	28.57	-28.57
72905	41-50	0	0	7	0.00	0.00	0.00
72058	1-10	4	4	7	57.14	57.14	0.00
72097	1-10	3	3	7	42.86	42.86	0.00
72048	41-50	5	2	7	71.43	28.57	-42.86
72189	21-30	3	3	7	42.86	42.86	0.00

GP – Guild Presence
PGP – Percent Guild Presence
Adj_No – is maximum number of guild members along any route in Pennsylvania

Table 4.2b – Breeding Bird Survey data for response guilds forest interior habitat birds and large forest obligate birds for each survey route. The change column reports the percent change between the two dates.

Forest Interior Guild – 46 Members

ROTNO	STOPS	GP_9193	GP_9901	Adj_No	PGP_92	PGP_01	Change
72038	1-10	9	10	31	29.03	32.26	3.23
72031	41-50	10	9	31	32.26	29.03	-3.23
72901	11-20	6	10	31	19.35	32.26	12.91
72014	41-50	15	18	31	48.39	58.06	9.67
72006	41-50	12	16	31	38.71	51.61	12.90
72017	31-40	13	13	31	41.94	41.94	0.00
72036	41-50	5	4	31	16.13	12.90	-3.23
72181	31-40	2	3	31	6.45	9.68	3.23
72009	41-50	10	20	31	32.26	64.52	32.26
72080	11-20	0	1	31	0.00	3.23	3.23
72905	41-50	10	12	31	32.26	38.71	6.45
72058	1-10	10	3	31	32.26	9.68	-22.58
72097	1-10	4	3	31	12.90	9.68	-3.22
72048	41-50	6	10	31	19.35	32.26	12.91
72189	21-30	17	17	31	54.84	54.84	0.00

Large Forest Obligate Guild – 23 Members

ROTNO	STOPS	GP_9193	GP_9901	Adj_No	PGP_92	PGP_01	Change
72038	1-10	5	6	17	29.41	35.29	5.88
72031	41-50	4	4	17	23.53	23.53	0.00
72901	11-20	1	3	17	5.88	17.65	11.77
72014	41-50	7	10	17	41.18	58.82	17.64
72006	41-50	8	10	17	47.06	58.82	11.76
72017	31-40	8	5	17	47.06	29.41	-17.65
72036	41-50	0	1	17	0.00	5.88	5.88
72181	31-40	0	1	17	0.00	5.88	5.88
72009	41-50	5	9	17	29.41	52.94	23.53
72080	11-20	0	1	17	0.00	5.88	5.88
72905	41-50	7	6	17	41.18	35.29	-5.89
72058	1-10	4	0	17	23.53	0.00	-23.53
72097	1-10	1	2	17	5.88	11.76	5.88
72048	41-50	3	5	17	17.65	29.41	11.76
72189	21-30	11	12	17	64.71	70.59	5.88

GP – Guild Presence
PGP – Percent Guild Presence
Adj_No – is maximum number of guild members along any route in Pennsylvania

Table 4.3 – Best subset variable selection results for each avian response guild.

Guild	MPS	AWFD	MNN	CORE	EDGE	VENTE	GRAND	MED	PEQ
FOR_OB	X	-	-	X	-	-	X	X	X
FOR-INT	X	-	-	X	-	X	X	-	-
GAS	-	X	-	X	X	-	X	-	-
GRASL	X	X	-	-	-	-	-	X	X

Variable list:

MPS	Mean Forest Patch Size
AWFD	Area Weighted Mean Fractal Dimension
MNN	Mean Nearest Neighbor
CORE	Core Forest
EDGE	Edge Forest
VENTE	Largest Forest Patch Size Group (>= 1000 ha)
GRAND	Large Forest Patch Size Group (< 1000 & >= 100 ha)
MED	Medium Forest Patch Size Group (< 100 & >= 10 ha)
PEQ	Small Forest Patch Size Group (< 10 ha)

Table 4.4 – Multiple linear regression results for each response guild. Also, the single best variable and its R^2-value and P-value are included.

Guild	Num.	R-Sq	P-Value	SV	R-Sq	P-Value
FOR_OB	5	53.7%	0.160	Grande	14.5%	0.161
FOR_INT	4	36.0%	0.626	MED_CH	11.8%	0.211
GAS	5	69.1%	0.013	MED_CH	32.7%	0.026
GRASL	4	44.0%	0.176	PEQ_CH	16.7%	0.130

Variable list:

Guild	Abbreviated guild name
Num	Number of variables includes in the linear model.
R-Sq(a)	R^2 adjusted – percent of variation in response variable explained by the independent variables.
P-Value	probability the result would occur randomly.
SV	Single best variable

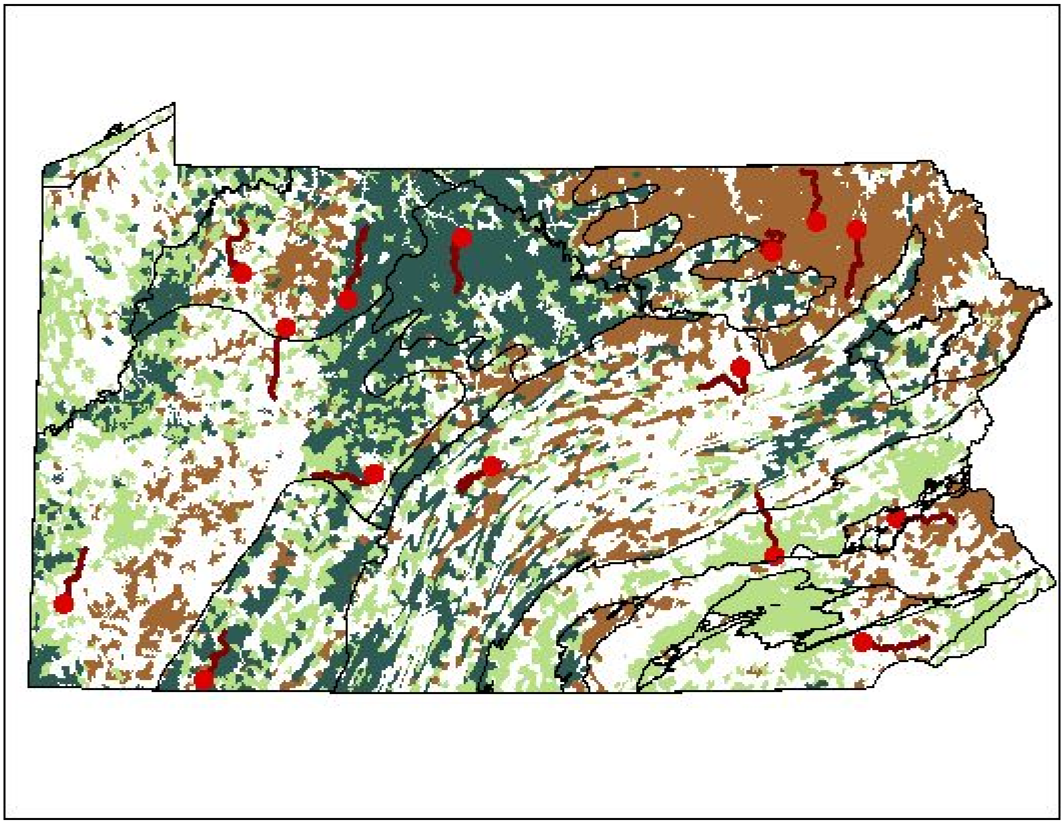


Figure 4.1 – Selected breeding bird survey routes where guild data were tallied for this study. Red lines represent each BBS route and the red dot the route's start point. Background colors show changes in percent core forest tabulated within landtype association (LTA) boundaries. Dark-green indicates an increase in core forest where brown represents a decrease. Light-green is a small increase and white is no change. Black lines delineate ecoregions of Pennsylvania.

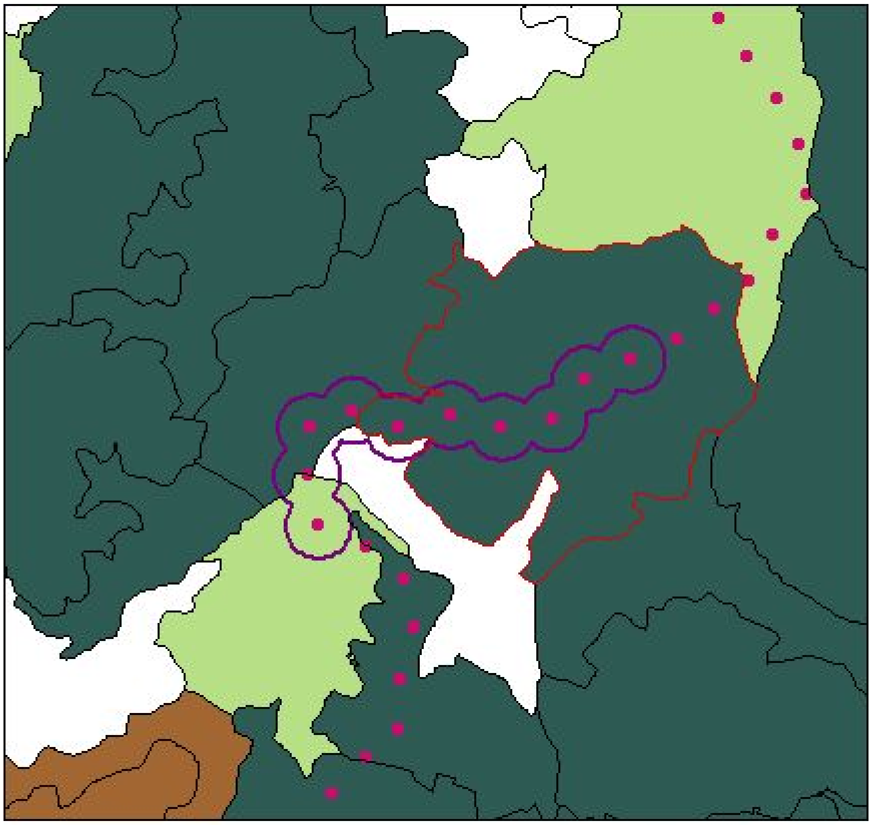
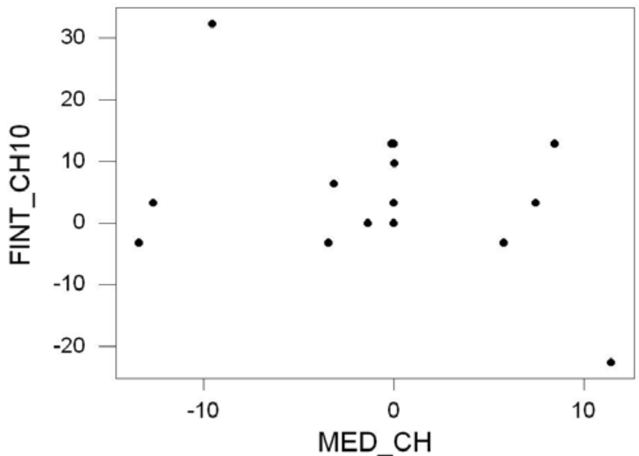


Figure 4.2 – Close-up of Figure (4.1). Depicts an individual breeding bird survey route where guild data were tallied during this study. The red dots represent BBS survey points and outlined in purple is the 10-stop segment selected from this route. The red boundary identifies the LTA polygon where fragmentation metrics were extracted for comparison to guild presence. Background colors show changes in percent core forest tabulated within landtype association (LTA) boundaries. Dark-green indicates an increase in core forest where as brown represents a decrease. Light-green is a small increase and white is no change.



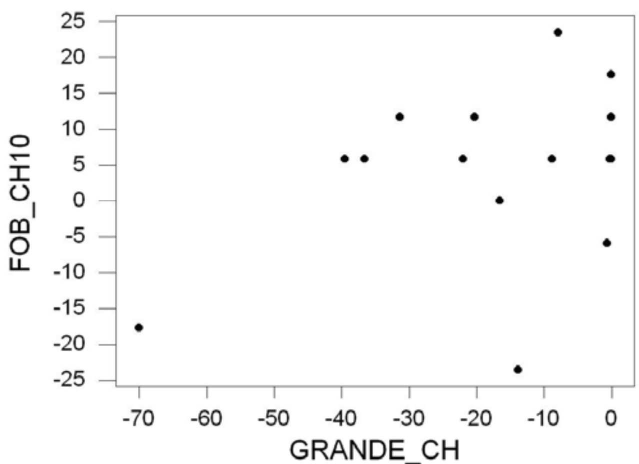


Figure 4.3a – Sample scatter plots for the forest interior guild (top) and the large forest obligate guild (bottom). The trend in the top graph shows the guild presence decreasing as the composition of the medium forest patch size group (10, 25, and 50 ha) increases. The bottom graph shows that guild presence increases as the large GRANDE forest patch size group (100, 250, and 500 ha) composition increases. To view all variables see matrix plots in Appendix E.

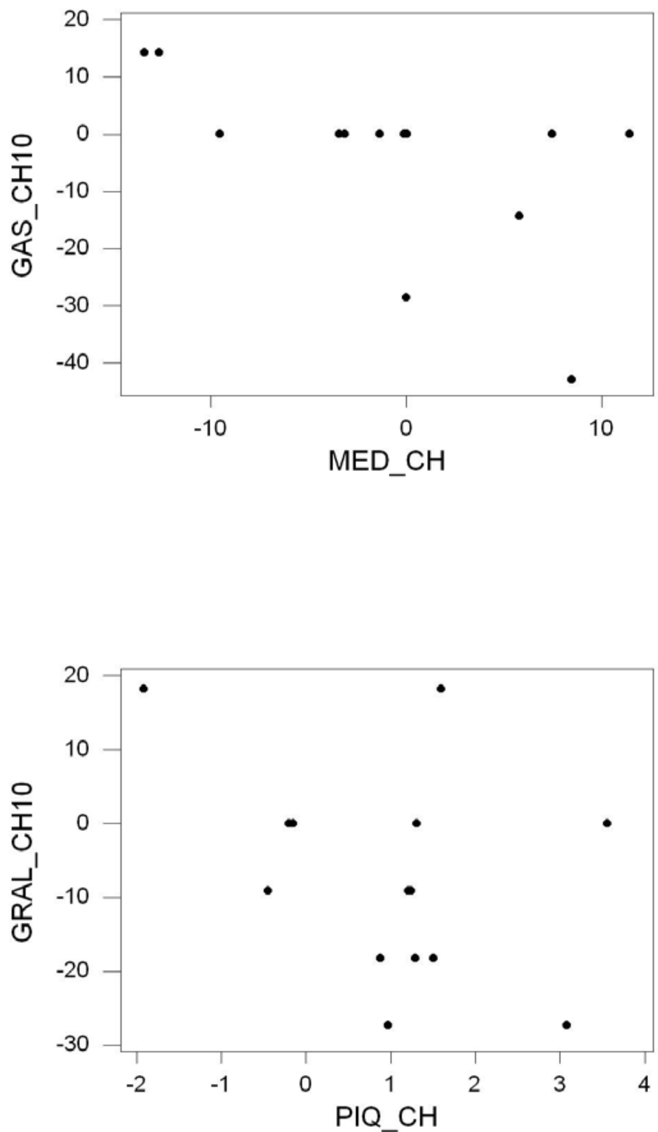


Figure 4.3b – Sample scatter plots for the grassland area sensitive guild (top) and the grassland habitat guild (bottom). The trend in the top graph shows the guild presence decreasing as the composition of the medium forest patch size group (10, 25, and 50 ha) increases. The bottom graph shows that guild presence decreases as the smallest forest patch size group (1, 2, and 5 ha) composition increases. To view all variables see matrix plots in Appendix E.

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Appendix

Appendix A.

1992 Land Cover Classification Summary (from Myers et al. 2000)

Generalized land cover and disturbance were mapped in several modes from Landsat Thematic Mapper (TM) digital image data collected during a period from 1991 through 1994. The Landsat image data were obtained from USGS EROS Data Center through the Multi-Resolution Land Characterization (MRLC) consortium. Image data were compressed through a hyperclustering protocol configured at Penn State Univ. for display and classification using commercial software. The compressed images have been made available to the public and have received considerable use in Pennsylvania as backdrops for GIS applications. An initial binary classification into *naturalistic* and *humanisitic* types of landscape matrix at 100-ha resolution was done by interactively digitizing with a mouse on a computer display. An interpretive classification of disturbance was done similarly, but with no specific minimum resolution. By reference to digital orthophoto quarter-quads (DOQQs), clusters were interpretively assigned to 8 general physiognomic land cover classes. Combining land cover and disturbance yielded 24 map classes for habitat modeling and analysis.

<u>The Land Cover Classification Scheme:</u>

The Pennsylvania 100-ha binary landscape matrix layer has codes as follows:

10 = naturalistic (forest, water, wetlands);
20 = humanistic (transitional, perennial herbaceous, annual herbaceous, barren).

The Pennsylvania 2-ha land-cover/disturbance layer has a two-digit coding scheme, for which the first digit is coded as:

1 = Rural (wild land or agriculture);
2 = Suburban (primarily low-density residential);
3 = Urban (primarily high-density residential and/or commercial/industrial);

with the second digit being a code for physiognomy as follows:

1 = Open water or wetlands with standing water;
2 = Evergreen forest (not more than 30% of tree canopy cover deciduous);
3 = Mixed forest (deciduous and evergreen both > 30% of tree canopy cover);
4 = Deciduous forest (not more than 30% of tree canopy cover evergreen);
5 = Woody transitional (5%< cover of woody plant foliage<40%), also shrubland or forest regeneration;
6 = Perennial herbaceous (grasslands, pasture, forage, old fields <5% shrubs);
7 = Annual herbaceous (row crops, grain crops, exposed mineral soil);
8 = Barren, hard-surface, rubble, gravel.

These latter classes form a natural ordination not only for physiognomy, but also for near-infrared spectral brightness. Spectral confusion is more likely for classes that are adjacent in the ordination than for classes that are further apart. Additional levels of classification were considered in cooperation with other northeastern states, but could not be implemented consistently using available image data.

Imagery Used:

The primary source of remotely sensed image data used in land-cover/disturbance mapping was from the Landsat Thematic Mapper (TM) sensor in paths 14-18 and rows 31 & 32, with coverage as shown in Figure 2.1. The data for these images were acquired from USGS EROS Data Center through GAP participation in the MRLC (Multi-Resolution Land Characterization) consortium. Each frame consisted of six bands, not including the thermal infrared. The image dates obtained for these path/row positions are listed in Table 2.1. Delivery of the image tapes was considerably delayed, which became a major cause of protraction for the Pennsylvania GAP Project. Several of the image dates were also considerably less than ideal for land classification, being acquired in phenological circumstances when trees had only partial foliage or were devoid of foliage. Clouds in portions of several images also required substantial remedial effort.

Dates of Landsat TM imagery.

Path/Row	Date 1	Date 2
014/031	5/9/93	-------
014/032	3/17/91	5/20/91
015/031	6/14/92	10/7/93
015/032	10/20/92	6/17/93
016/031	5/20/92	-------
016/032	8/24/92	6/24/93
017/031	5/11/92	10/2/92
017/032	10/2/92	5/14/93
018/031	8/9/93	4/22/94
018/032	4/22/92	8/6/92

Appendix A.

2001 Land Cover Classification Summary (from Myers & Warner 2003)

Land cover information was generated from a combination of satellite and vector ancillary data. Imagery from the Enhanced Thematic Mapper (ETM) sensor, carried on the Landsat 7 platform, served as the primary data source for land cover interpretation. Ten ETM scenes cover Pennsylvania completely, with overlap into neighboring states. ETM scenes are referenced based on a global system whereby each location on earth has a unique path and row identification number. To provide spectral contrast between differing land covers, two ETM scenes were sought for each path and row area in the state. The scenes were chosen to capture conditions in early to mid-autumn and mid-summer. A few scenes were purchased outside these windows due to cloud cover. Cloudiness also made it necessary to acquire data over a three-year period. Path/Row and dates of the images used in the interpretation are as follows:

Path/Row	Date		
14/31	6/11/2002	14/31	09/12/2001
14/32	7/05/1999	14/32	09/12/2001
15/31	7/28/1999	15/31	11/10/2001
15/32	8/02/2001	15/32	11/17/1999
16/31	7/05/2000	16/31	08/25/2001
16/32	8/04/1999	16/32	09/07/2000
17/31	6/13/2001	17/31	09/17/2001
17/32	6/10/2000	17/32	09/17/2001
18/31	8/07/2001	18/31	09/08/2001
18/32	7/06/2001	18/32	09/08/2001

Typical of spectrally based land cover classifications, there was noticeable confusion between certain classes. To solve some of this ambiguity, ancillary data layers were incorporated into the land cover mapping process. Data layers incorporated into the land cover classification include: 1997 PA-GAP Urban Classification Layer and 1992 NLCD Classification.

Land Cover Mapping Process

1. All ETM images were classified using Dr. Wayne Myer's, Professor of Forestry, Penn State Univ., PSISCAN program into 250 groups, forming a single band image. The single band classified images were then used to create color renderings that inserted as backdrops during the visual interpretation process. The visual interpretation yielded land cover codes for each of the 250 groups comprising a single image. Coding including a single numeric code indicating the land cover most frequently represented by a group and a multi-integer code indicative of all land covers included in a group. Some of the 250 groups had only single integer codes when completely unique.

2. After the initial interpretation, land cover classifications were assembled for each path and row area based on codes from two dates of imagery. Comparisons were made between dates to determine which were best for groups or if combinations could be made to improve the overall classification. Having two dates of imagery also allowed for clouded areas to assume the code of the second date.

3. The coding in #2 was modified with the PA-GAP urban classification layer, which was used to classify areas into either low or high intensity uses. Locations that were classified as grass in the initial classification were left that way, as fields, parks and large lawns commonly occur in urban areas.

4. Once the urban classification was complete, north/south neighboring image areas (image areas with the same path but different row) were merged. These merged layers were compared with neighboring land cover layers to identify discrepancies in the classification process.

5. The 1994 NLCD interpretation was used to assign land cover codes in four situations; 1) Areas that were cloudy during both image dates for a row/path location, were coded from the 1994 NLCD data. Areas cloudy in both images comprised only 0.000001% of the land area in the state. 2) The wetland groups were taken from the NLCD data. The classification of wetlands is not very reliable using spectral information only. Further based on the regulatory goal of zero wetland loss, the area of wetlands should be very close to that seen in 1994. 3) Quarry areas and coal mines were assigned from the 1994 NLCD as their extents have seen little changed. Coal mine boundaries could have been assigned from data available from the Bureau of Mines. It was decided not to use the information as comparison of the NLCD and mine boundaries found poor agreement. Had the polygon information been used, comparison of the NLCD and the new land cover would have generated misleading statistics about the status of mining in Pennsylvania. 4) Locations identified as water in the NLCD data were coded water in the new image. Consistency in the mapping of water removes unlikely land conversion that would be identified through comparison of the 1994 and 2001 land cover data sets. Differences in mapping water can result from slight mis-registration errors, changes in water level, and confusion with other land cover groups. Final raster output from the processing includes 15 categories selected to match NLCD coding.

6. The cell value and the associated code are listed below;

1 - Water
2 - Low Density Urban
3 - High Density Urban
4 - Hay/Pasture
5 - Row Crops
6 - Probable Row Crops
7 - Coniferous Forest
8 - Mixed Forest
9 - Deciduous Forest
10 - Woody Wetland
11 - Emergent Wetland
12 - Quarries
13 - Coal Mines
14 - Beach
15 - Transitional

Appendix B.

Below is the expanded Table 2.3 “Accuracy assessment for physiognomic land cover categories” from Myers et al. (2000). R=reference; M=map. Water (watr), evergreen forest (evrgrn), mixed forest (mixd), deciduous forest (decid), transitional (trans), perennial herbaceous (pherb), annual herbaceous (aherb), barren (bare). Producer’s accuracy (%pac); user's accuracy (%uac). The next table summarizes it focusing on the Humanistic and Naturalist groupings.

LCov Class	Rwatr	Revrgrn	Rmixd	Rdecid	Rtrans	Rpherb	Raherb	Rbare	Rtotal	%Uac
Mwatr	40	0	1	8	1	0	3	2	55	72.7
Mevrgrn	0	3	3	3	1	1	0	1	12	25
Mmixd	1	8	47	13	8	1	2	3	83	56.6
Mdecid	0	3	5	284	16	9	9	7	333	85.3
Mtrans	3	0	1	5	18	2	3	2	34	52.9
Mpherb	1	2	4	20	9	42	20	5	103	40.8
Maherb	0	1	2	6	5	5	68	3	90	75.6
Mbare	2	4	4	5	3	3	3	21	45	46.7
Mtotal	47	21	67	344	61	63	108	44	755	
%Pac	85.1	14.2	70.1	82.6	29.5	66.7	63	47.7		69.3

Nat/Hum	Rwater	Rhuman	Rnatural	Count	Percent
Mwater	40.0	5.0	10.0	55.0	72.7
Mhuman	3.0	170.0	65.0	238.0	7.1
Mnatural	4.0	40.0	418.0	462.0	90.5
Count	47.0	215.0	493.0	**755.0**	
Percent	85.1	79.1	84.8		83.2

Appendix B.

Below is the expanded Table 2.4, a detailed validation of the 2001 land cover. Each table illustrates the effect of a different method of validating the same data. Raw is the exact pixel-to-pixel most conservative view. Pooled reflects the grouping of related land cover classes (3 forest classes together, 2 developed & 2 grassland/agriculture). Circle looks at the composition of a small 3x3 region around each point. Cir-Pool combined the circle and pooled method. R=reference; M=map. Water (water), low intensity development (LowDev), high intensity development (HighDev), pasture (Pasture), row crop (RowCrop), coniferous forest (Conifer), mixed forest (Mix), deciduous forest (Decidu), transitional (Trans). Producer’s accuracy (%pac); user's accuracy (%uac). The next table summarizes it focusing on the Humanistic and Naturalist groupings.

Raw	RWater	RLowDev	RHighDev	Rpasture	RRowCrop	Rconifer	Rmix	Rdecidu	Rtrans	Count	%UAC
Mwater	16.0	0.0	1.0	0.0	1.0	0.0	0.0	1.0	0.0	19.0	84.2
MLowDev	0.0	5.0	0.0	0.0	1.0	0.0	0.0	2.0	0.0	8.0	50.0
MHighDev	0.0	5.0	5.0	0.0	0.0	0.0	0.0	1.0	0.0	11.0	50.0
Mpasture	0.0	1.0	0.0	6.0	1.0	0.0	2.0	0.0	2.0	12.0	50.0
MRowCrop	0.0	2.0	2.0	9.0	22.0	1.0	0.0	5.0	3.0	44.0	53.5
Mconifer	0.0	0.0	0.0	0.0	0.0	1.0	1.0	9.0	1.0	12.0	8.3
Mmix	0.0	0.0	2.0	1.0	0.0	2.0	2.0	7.0	1.0	15.0	14.3
Mdecidu	4.0	0.0	1.0	3.0	1.0	3.0	5.0	59.0	8.0	84.0	64.9
Mtrans	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	1.0	0.0
Count	20.0	13.0	11.0	19.0	26.0	7.0	10.0	85.0	15.0	**206.0**	
%PAC	80.0	38.5	45.5	31.6	84.6	14.2	20.0	69.4	0.0		

Pooled	RWater	RLowDev	RHighDev	Rpasture	RRowCrop	Rconifer	Rmix	Rdecidu	Rtrans	Count	%UAC
Mwater	16.0	0.0	1.0	0.0	1.0	0.0	0.0	1.0	0.0	17.0	94.1
MLowDev	0.0	10.0	0.0	0.0	1.0	0.0	0.0	2.0	0.0	13.0	70.0
MHighDev	0.0	0.0	5.0	0.0	0.0	0.0	0.0	1.0	0.0	6.0	83.3
Mpasture	0.0	1.0	0.0	15.0	0.0	0.0	2.0	0.0	2.0	20.0	72.2
MRowCrop	0.0	2.0	2.0	0.0	23.0	1.0	0.0	5.0	3.0	36.0	65.7
Mconifer	0.0	0.0	0.0	0.0	0.0	6.0	0.0	0.0	1.0	7.0	85.7
Mmix	0.0	0.0	2.0	1.0	0.0	0.0	8.0	0.0	1.0	12.0	66.7
Mdecidu	4.0	0.0	1.0	3.0	1.0	0.0	0.0	75.0	8.0	94.0	76.2
Mtrans	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	1.0	0.0
Count	20.0	13.0	11.0	19.0	26.0	7.0	10.0	85.0	15.0	**206.0**	
%PAC	80.0	76.9	45.5	78.9	88.5	85.7	80.0	88.2	0.0		

Circle	RWater	RLowDev	RHighDev	Rpasture	RRowCrop	Rconifer	Rmix	Rdecidu	Rtrans	Count	%UAC
Mwater	18.0	0.0	1.0	0.0	1.0	0.0	0.0	1.0	0.0	21.0	85.7
MLowDev	0.0	8.0	0.0	0.0	1.0	0.0	0.0	2.0	0.0	11.0	72.7
MHighDev	0.0	3.0	5.0	0.0	0.0	0.0	0.0	1.0	0.0	9.0	55.6
Mpasture	0.0	1.0	0.0	9.0	0.0	0.0	1.0	0.0	2.0	13.0	69.2
MRowCrop	0.0	1.0	2.0	7.0	23.0	0.0	0.0	4.0	3.0	40.0	57.5
Mconifer	0.0	0.0	0.0	0.0	0.0	3.0	1.0	4.0	1.0	9.0	33.3
Mmix	0.0	0.0	2.0	0.0	0.0	2.0	3.0	4.0	1.0	12.0	25.0
Mdecidu	2.0	0.0	1.0	3.0	1.0	2.0	5.0	69.0	8.0	91.0	75.8
Mtrans	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Count	20.0	13.0	11.0	19.0	26.0	7.0	10.0	85.0	15.0	**206.0**	
%PAC	90.0	61.5	45.5	47.4	88.5	42.9	30.0	81.2	0.0		

Cir_Pool	RWater	RLowDev	RHighDev	Rpasture	RRowCrop	Rconifer	Rmix	Rdecidu	Rtrans	Count	%UAC
Mwater	18.0	0.0	1.0	0.0	1.0	0.0	0.0	1.0	0.0	21.0	85.7
MLowDev	0.0	8.0	0.0	0.0	1.0	0.0	0.0	2.0	0.0	11.0	72.7
MHighDev	0.0	0.0	8.0	0.0	0.0	0.0	0.0	1.0	0.0	9.0	88.9
Mpasture	0.0	1.0	0.0	9.0	0.0	0.0	1.0	0.0	2.0	13.0	69.2
MRowCrop	0.0	1.0	2.0	0.0	30.0	0.0	0.0	4.0	3.0	40.0	75.0
Mconifer	0.0	0.0	0.0	0.0	0.0	8.0	0.0	0.0	1.0	9.0	88.9
Mmix	0.0	0.0	2.0	0.0	0.0	0.0	9.0	0.0	1.0	12.0	75.0
Mdecidu	2.0	0.0	1.0	3.0	1.0	0.0	0.0	76.0	8.0	91.0	83.5
Mtrans	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Count	20.0	11.0	14.0	12.0	33.0	8.0	10.0	83.0	15.0	**206.0**	
%PAC	90.0	72.7	57.1	75.0	90.9	100.0	90.0	91.6	0.0		

Appendix C. - Response Guilds

* - Indicates State listed or PIF watch list bird.

Guild - Grassland Area Sensitive

AOU Code	Scientific Name	Common Name
HOHA	*Circus cyaneus*	Northern harrier
UPSA*	*Bartramia longicauda*	Upland sandpiper
BNOW*	*Tyto alba*	Barn owl
SEOW	*Asio flammeus*	Short-eared owl
HOLA	*Eremophila alpestris*	Horned lark
VESP*	*Pooecetes gramineus*	Vesper sparrow
SASP	*Passerculus sandwichensis*	Savannah sparrow
GRSP*	*Ammodramus savannarun*	Grasshopper sparrow
HESP*	*Ammodramus henslowii*	Henslow's sparrow
BOBO	*Dolichonyx orizyvorus*	Bobolink
EAME	*Sturnella magna*	Eastern meadowlark

Guild - Grassland Habitat

AOU Code	Scientific Name	Common Name
HOHA*	*Circus cyaneus*	Northern harrier
AMKE	*Falco sparverius*	American kestrel
RNPH	*Phasianus colchicus*	Ring-necked pheasant
NOBO	*Colinus virginianus*	Northern bobwhite
KILL	*Charadrius vociferus*	Killdeer
UPSA*	*Bartramia longicauda*	Upland sandpiper
BNOW*	*Tyto alba*	Barn owl
SEOW*	*Asio flammeus*	Short-eared owl
CONI	*Chordeiles minor*	Common nighthawk
HOLA	*Eremophila alpestris*	Horned lark
BANS	*Riparia riparia*	Bank swallow
LOSH	*Lanius ludovicianus*	Loggerhead shrike
DICK*	*Spiza americana*	Dickcissel
CCSP*	*Spizella pallida*	Clay-colored sparrow
VESP*	*Pooecetes gramineus*	Vesper sparrow
SASP	*Passerculus sandwichensis*	Savannah sparrow
GRSP*	*Ammodramus savannarun*	Grasshopper sparrow
HESP*	*Ammodramus henslowii*	Henslow's sparrow
BOBO	*Dolichonyx orizyvorus*	Bobolink
EAME	*Sturnella magna*	Eastern meadowlark
WEME	*Sturnella neglecta*	Western meadowlark

Guild - Forest Interior Habitat

AOU Code	Scientific Name	Common Name
SSHA	*Accipiter striatus*	Sharp-shinned hawk
COHA	*Accipiter cooperii*	Cooper's hawk
NOGO*	*Accipiter gentilis*	Northern goshawk
RSHA*	*Buteo lineatus*	Red-shouldered hawk
BWHA	*Buteo platypterus*	Broad-winged hawk
WITU	*Meleagris gallopavo*	Wild turkey
BAOW	*Strix varia*	Barred owl
LEOW	*Asio otus*	Long-eared owl
NSWO	*Aegolius acadicus*	Northern saw-whet owl
CWWI*	*Caprimulgus carolinensis*	Chuck Will's widow
WPWI*	*Caprimulgus vociferus*	Whip-poor-Will
HAWO	*Picoides villosus*	Hairy woodpecker
PIWO	*Dryocopus pileatus*	Pileated woodpecker
OSFL*	*Contopus borealis*	Olive-sided flycatcher
YBFL	*Empidonax flaviventris*	Yellow-bellied flycatcher
ACFL*	*Empidonax virescens*	Acadian flycatcher
CORA	*Corvus corax*	Common raven
RBNU	*Sitta canadensis*	Red-breated nuthatch
WBNU	*Sitta carolinensis*	White-breasted nuthatch
BRCR	*Certhia americana*	Brown creeper
WIWR	*Troglodytes troglodytes*	Winter wren
GCKI	*Regulus satrapa*	Golden-crowned kinglet
VEER	*Catharus fuscescens*	Veery
HETH*	*Catharus guttatus*	Hermit thrush
BHVI	*Vireo solitarius*	Blue-headed vireo
OCWA	*Vermivora celata*	Orange-crowned warbler
MAWA*	*Dendroica magnolia*	Magnolia warbler
BTBW	*Dendroica caerulescens*	Black-throated blue warbler
YRWA	*Dendroica coronata*	Yellow-rumped warbler
BTGW*	*Dendroica virens*	Black-throated green warbler
BLBA*	*Dendroica fusca*	Blackburnian warbler
YTWA*	*Dendroica dominica*	Yellow-throated warbler
PIWA*	*Dendroica pinus*	Pine warbler
CERW*	*Dendroica cerulea*	Cerulean warbler
BAWW	*Mniotilta varia*	Black-and-white warbler
AMRE	*Setophaga ruticilla*	American redstart
WEWA*	*Helmitheros vermivorus*	Worm-eating warbler
SWWA*	*Limnothlypis swainsonii*	Swainson's warbler
OVEN	*Seiurus aurocapillus*	Ovenbird
NOWA	*Seiurus noveboracensis*	Northern waterthrush
LOWA*	*Seiurus motacilla*	Louisiana waterthrush
KEWA*	*Oporornis formosus*	Kentucky warbler
HOWA*	*Wilsonia citrina*	Hooded warbler
CAWA*	*Wilsonia canadensis*	Canada warbler
SCTA*	*Piranga olivacea*	Scarlet tanager
PISI	*Carduelis pinus*	Pine siskin

Guild - Large Forest Obligate

AOU Code	Scientific Name	Common Name
SSHA	*Accipiter striatus*	Sharp-shinned Hawk
NOGO*	*Accipiter gentilis*	Northern goshawk
BAOW	*Strix varia*	Barred owl
YBFL	*Empidonax flaviventris*	Yellow-bellied Flycatcher
ACFL*	*Empidonax virescens*	Acadian flycatcher
BRCR*	*Certhia americana*	Brown creeper
WIWR	*Troglodytes troglodytes*	Winter wren
VEER	*Catharus fuscescens*	Veery
SWTH	*Catharus ustulatus*	Swainson's Thrush
HETH*	*Catharus guttatus*	Hermit thrush
BHVI	*Vireo solitarius*	Blue-headed vireo
NOPA	*Parula americana*	Northern Parula
MAWA*	*Dendroica magnolia*	Magnolia Warbler
BTBW	*Dendroica caerulescens*	Black-throated blue warbler
BTGW*	*Dendroica virens*	Black-throated green warbler
BLBA*	*Dendroica fusca*	Blackburnian Warbler
YTWA*	*Dendroica pinus*	Yellow-throated Warbler
CERW*	*Dendroica cerulea*	Cerulean warbler
BAWW	*Mniotilta varia*	Black-and-white warbler
WEWA*	*Helmitheros vermivorus*	Worm-eating warbler
KEWA*	*Oporornis formosus*	Kentucky warbler
HOWA*	*Wilsonia citrina*	Hooded warbler
CAWA*	*Wilsonia canadensis*	Canada warbler

* - Indicates State listed or PIF watch list bird. Based on (Brauning et al. 1994; R. Blye, personal communication 2003; Rich et al. 2004)

Appendix D.

Complete metric change tables of each response guild. The values on the left represent all change metrics that were tested for each BBS route. The values on the right are the guild counts for that route.

Large Forest Obligates

ROTNO	MPS	AWFD	MNN	CORE	EDGE	VENTE	GRANDE	MED	PEQ	ST	GP92	GP01	ADJ	PGP92	PGP01	FOB_CH
72038	-27.240	0.040	7.670	-34.15	6.26	0.00	-36.66	7.48	1.31	1	5	6	17	29.41	35.29	5.88
72031	-60.310	0.020	13.930	-17.71	4.02	-4.00	-16.52	5.81	1.24	5	4	4	17	23.53	23.53	0.00
72901	-10.700	0.020	19.900	-16.68	-2.42	0.00	-20.23	-0.09	1.21	2	1	3	17	5.88	17.65	11.77
72014	104.790	-0.040	-4.990	21.30	-12.77	8.67	0.00	0.03	-0.15	5	7	10	17	41.18	58.82	17.64
72006	3.340	0.000	-4.280	0.23	1.82	34.10	-31.40	0.00	-0.45	5	8	10	17	47.06	58.82	11.76
72017	96.130	-0.050	0.000	16.08	-6.88	79.49	-70.10	0.00	-0.20	4	8	5	17	47.06	29.41	-17.65
72036	-9.710	0.000	8.120	-3.61	-6.50	0.00	-0.22	-13.44	3.56	5	0	1	17	0.00	5.88	5.88
72181	-8.320	0.020	9.870	-11.82	-6.50	0.00	-8.83	-12.67	3.08	4	0	1	17	0.00	5.88	5.88
72009	-4.940	0.000	25.110	-8.55	-7.29	0.00	-7.87	-9.56	1.60	5	5	9	17	29.41	52.94	23.53
72080	-3.430	-0.040	159.110	0.00	0.88	0.00	0.00	0.00	0.88	2	0	1	17	0.00	5.88	5.88
72905	67.470	-0.040	-8.490	17.21	-5.93	15.28	-0.64	-3.16	-0.20	5	7	6	17	41.18	35.29	-5.89
72058	-11.930	-0.010	-0.250	-3.68	0.92	-1.87	-13.86	11.47	1.51	1	4	0	17	23.53	0.00	-23.53
72097	-11.720	-0.010	64.490	-23.35	-18.31	0.00	-39.50	-3.45	1.29	1	1	2	17	5.88	11.76	5.88
72048	-2.190	-0.020	0.070	3.18	6.29	0.00	0.00	8.50	0.97	5	3	5	17	17.65	29.41	11.76
72189	15.540	0.010	-6.350	8.04	2.89	36.39	-21.97	-1.37	-1.92	3	11	12	17	64.71	70.59	5.88

Forest Interior

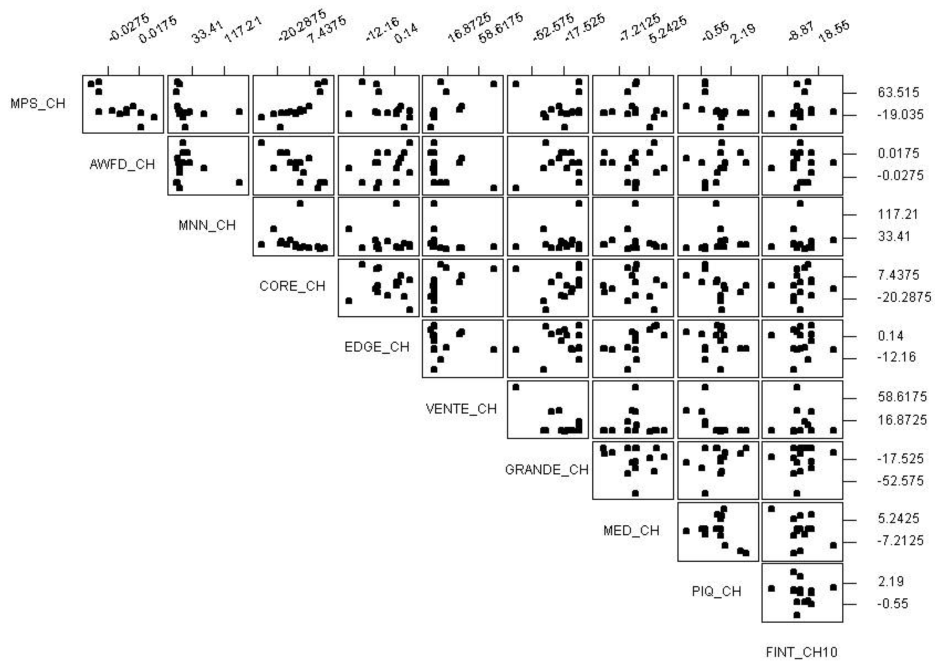
ROTNO	MPS	AWFD	MNN	CORE	EDGE	VENTE	GRANDE	MED	PEQ	ST	GP92	GP01	ADJ	PGP92	PGP01	FINT_CH
72038	-27.240	0.040	7.670	-34.15	6.26	0.00	-36.66	7.48	1.31	1	9	10	31	29.03	32.26	3.23
72031	-60.310	0.020	13.930	-17.71	4.02	-4.00	-16.52	5.81	1.24	5	10	9	31	32.26	29.03	-3.23
72901	-10.700	0.020	19.900	-16.68	-2.42	0.00	-20.23	-0.09	1.21	2	6	10	31	19.35	32.26	12.91
72014	104.790	-0.040	-4.990	21.30	-12.77	8.67	0.00	0.03	-0.15	5	15	18	31	48.39	58.06	9.67
72006	3.340	0.000	-4.280	0.23	1.82	34.10	-31.40	0.00	-0.45	5	12	16	31	38.71	51.61	12.90
72017	96.130	-0.050	0.000	16.08	-6.88	79.49	-70.10	0.00	-0.20	4	13	13	31	41.94	41.94	0.00
72036	-9.710	0.000	8.120	-3.61	-6.50	0.00	-0.22	-13.44	3.56	5	5	4	31	16.13	12.90	-3.23
72181	-8.320	0.020	9.870	-11.82	-6.50	0.00	-8.83	-12.67	3.08	4	2	3	31	6.45	9.68	3.23
72009	-4.940	0.000	25.110	-8.55	-7.29	0.00	-7.87	-9.56	1.60	5	10	20	31	32.26	64.52	32.26
72080	-3.430	-0.040	159.110	0.00	0.88	0.00	0.00	0.00	0.88	2	0	1	31	0.00	3.23	3.23
72905	67.470	-0.040	-8.490	17.21	-5.93	15.28	-0.64	-3.16	-0.20	5	10	12	31	32.26	38.71	6.45
72058	-11.930	-0.010	-0.250	-3.68	0.92	-1.87	-13.86	11.47	1.51	1	10	3	31	32.26	9.68	-22.58
72097	-11.720	-0.010	64.490	-23.35	-18.31	0.00	-39.50	-3.45	1.29	1	4	3	31	12.90	9.68	-3.22
72048	-2.190	-0.020	0.070	3.18	6.29	0.00	0.00	8.50	0.97	5	6	10	31	19.35	32.26	12.91
72189	15.540	0.010	-6.350	8.04	2.89	36.39	-21.97	-1.37	-1.92	3	17	17	31	54.84	54.84	0.00

Grassland Area Sensitive

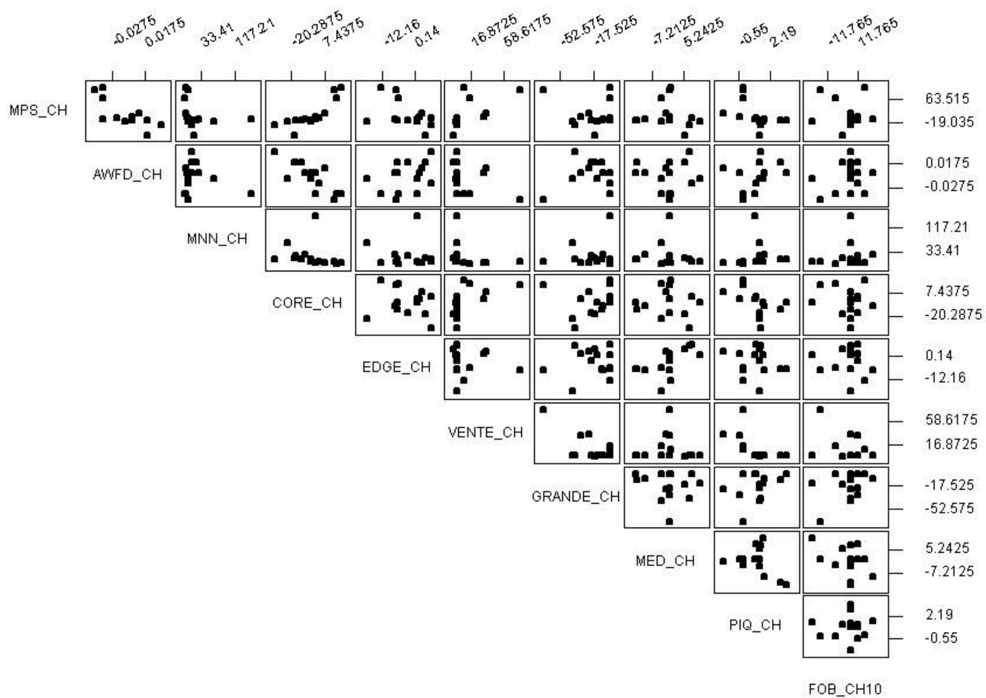
ROTNO	MPS	AWFD	MNN	CORE	EDGE	VENTE	GRANDE	MED	PEQ	ST	GP92	GP01	ADJ	PGP92	PGP01	GAS_CH10
72038	-27.240	0.040	7.670	-34.15	6.26	0.00	-36.66	7.48	1.31	1	2	2	7	28.57	28.57	0.00
72031	-60.310	0.020	13.930	-17.71	4.02	-4.00	-16.52	5.81	1.24	5	3	2	7	42.86	28.57	-14.29
72901	-10.700	0.020	19.900	-16.68	-2.42	0.00	-20.23	-0.09	1.21	2	3	3	7	42.86	42.86	0.00
72014	104.790	-0.040	-4.990	21.30	-12.77	8.67	0.00	0.03	-0.15	5	0	0	7	0.00	0.00	0.00
72006	3.340	0.000	-4.280	0.23	1.82	34.10	-31.40	0.00	-0.45	5	2	2	7	28.57	28.57	0.00
72017	96.130	-0.050	0.000	16.08	-6.88	79.49	-70.10	0.00	-0.20	4	0	0	7	0.00	0.00	0.00
72036	-9.710	0.000	8.120	-3.61	-6.50	0.00	-0.22	-13.44	3.56	5	4	5	7	57.14	71.43	14.29
72181	-8.320	0.020	9.870	-11.82	-6.50	0.00	-8.83	-12.67	3.08	4	1	2	7	14.29	28.57	14.28
72009	-4.940	0.000	25.110	-8.55	-7.29	0.00	-7.87	-9.56	1.60	5	4	4	7	57.14	57.14	0.00
72080	-3.430	-0.040	159.110	0.00	0.88	0.00	0.00	0.00	0.88	2	4	2	7	57.14	28.57	-28.57
72905	67.470	-0.040	-8.490	17.21	-5.93	15.28	-0.64	-3.16	-0.20	5	0	0	7	0.00	0.00	0.00
72058	-11.930	-0.010	-0.250	-3.68	0.92	-1.87	-13.86	11.47	1.51	1	4	4	7	57.14	57.14	0.00
72097	-11.720	-0.010	64.490	-23.35	-18.31	0.00	-39.50	-3.45	1.29	1	3	3	7	42.86	42.86	0.00
72048	-2.190	-0.020	0.070	3.18	6.29	0.00	0.00	8.50	0.97	5	5	2	7	71.43	28.57	-42.86
72189	15.540	0.010	-6.350	8.04	2.89	36.39	-21.97	-1.37	-1.92	3	3	3	7	42.86	42.86	0.00

Grassland Habitat

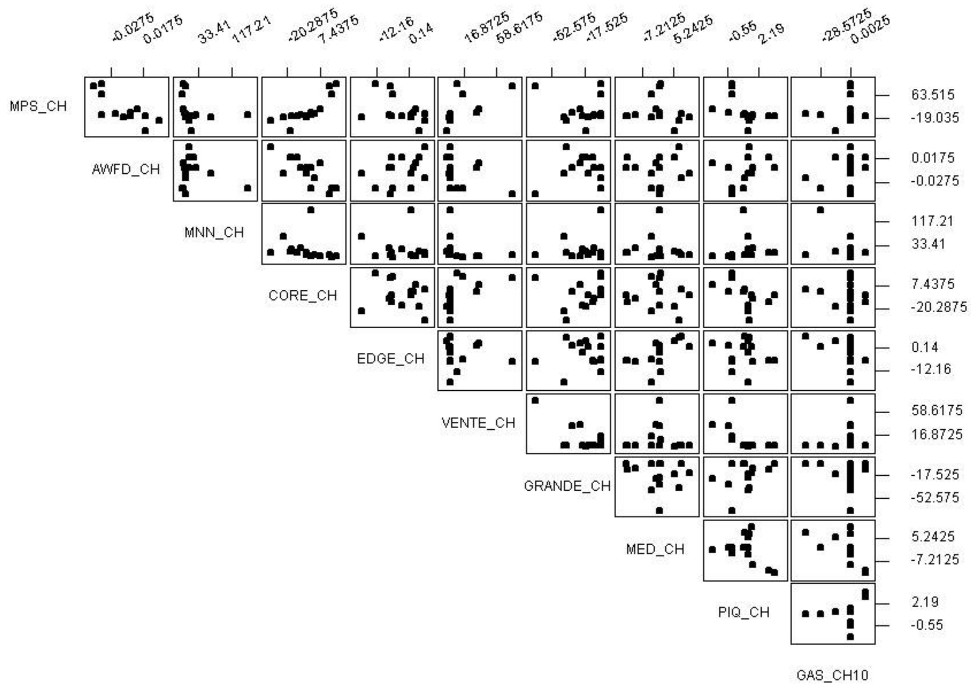
MPS	AWFD	MNN	CORE	EDGE	VENTE	GRANDE	MED	PEQ	ST	GP92	GP01	ADJ	PGP92	PGP01	GRAL_CH
-27.240	0.040	7.670	-34.15	6.26	0.00	-36.66	7.48	1.31	1	4	4	11	36.36	36.36	0.00
-60.310	0.020	13.930	-17.71	4.02	-4.00	-16.52	5.81	1.24	5	4	3	11	36.36	27.27	-9.09
-10.700	0.020	19.900	-16.68	-2.42	0.00	-20.23	-0.09	1.21	2	5	4	11	45.45	36.36	-9.09
104.790	-0.040	-4.990	21.30	-12.77	8.67	0.00	0.03	-0.15	5	0	0	11	0.00	0.00	0.00
3.340	0.000	-4.280	0.23	1.82	34.10	-31.40	0.00	-0.45	5	3	2	11	27.27	18.18	-9.09
96.130	-0.050	0.000	16.08	-6.88	79.49	-70.10	0.00	-0.20	4	0	0	11	0.00	0.00	0.00
-9.710	0.000	8.120	-3.61	-6.50	0.00	-0.22	-13.44	3.56	5	7	7	11	63.64	63.64	0.00
-8.320	0.020	9.870	-11.82	-6.50	0.00	-8.83	-12.67	3.08	4	4	1	11	36.36	9.09	-27.27
-4.940	0.000	25.110	-8.55	-7.29	0.00	-7.87	-9.56	1.60	5	4	6	11	36.36	54.55	18.19
-3.430	-0.040	159.110	0.00	0.88	0.00	0.00	0.00	0.88	2	6	4	11	54.55	36.36	-18.19
67.470	-0.040	-8.490	17.21	-5.93	15.28	-0.64	-3.16	-0.20	5	0	0	11	0.00	0.00	0.00
-11.930	-0.010	-0.250	-3.68	0.92	-1.87	-13.86	11.47	1.51	1	6	4	11	54.55	36.36	-18.19
-11.720	-0.010	64.490	-23.35	-18.31	0.00	-39.50	-3.45	1.29	1	6	4	11	54.55	36.36	-18.19
-2.190	-0.020	0.070	3.18	6.29	0.00	0.00	8.50	0.97	5	8	5	11	72.73	45.45	-27.28
15.540	0.010	-6.350	8.04	2.89	36.39	-21.97	-1.37	-1.92	3	2	4	11	18.18	36.36	18.18



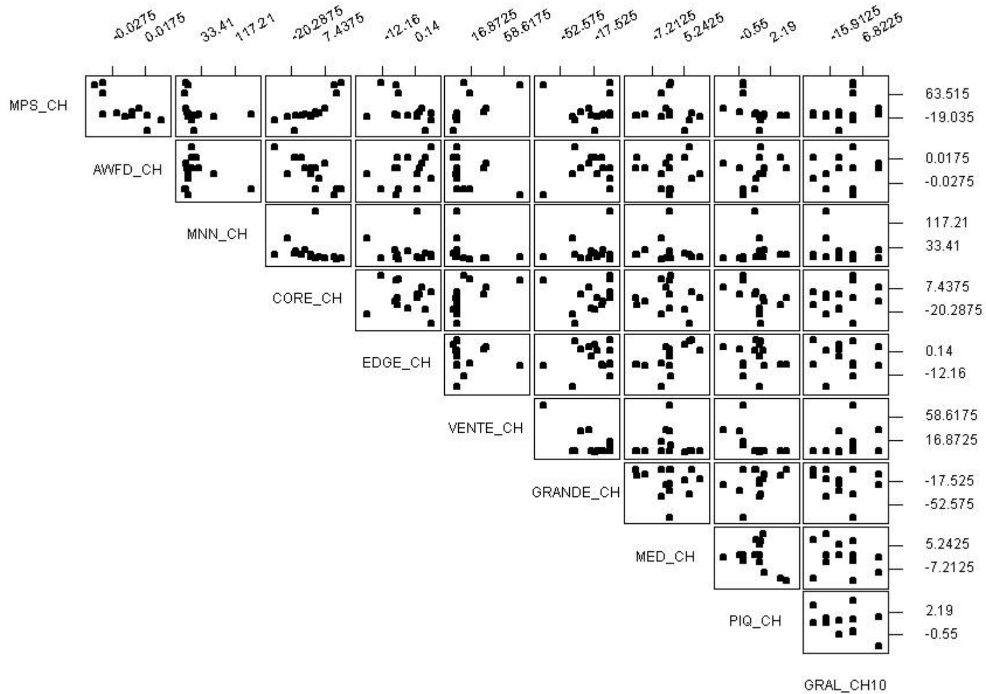
Appendix E. Matrix plot for the forest interior guild.



Appendix E. Matrix plot for the large forest obligate guild.



Appendix E. Matrix plot for the grassland area sensitive guild.



Appendix E. Matrix plot for the grassland habitat guild.

CURRICULUM VITAE

Joseph A. Bishop

Geospatial Coordinator, Penn State Cooperative Wetlands Center, Department of Geography &
Penn State Institute of Energy and the Environment

Penn State Cooperative Wetlands Center, Dept. of Geography, 302 Walker Bldg., Penn State University, University Park, PA, 16802
OFFICE: 814/863-3530; **CELL**: 814/876-3857; **FAX**: 814/865-3378; **EMAIL**: jab190@psu.edu

EDUCATION:

The Pennsylvania State University, University Park, PA, Ph.D. 2008, Ecology;
The Pennsylvania State University, University Park, PA, M.S. 2000, Ecology;
The Ohio State University, Columbus, OH, B.S. 1984, *with Honors* in Natural Resource Development.

PROFESSIONAL EXPERIENCE:

<u>2000-present</u> - Research Assistant, Penn State Cooperative Wetlands Center;
<u>1993-2000</u> – Graduate Research Assistant, Pennsylvania's Gap Analysis Project, Penn State;
<u>1991-1993</u> - Associate in Research, Organization for Tropical Studies, La Selva Biological Research Station, Costa Rica;
<u>1990-1991</u> - The Nature Conservancy, Connecticut Heritage Program, Hartford, Connecticut;
<u>1984-1989</u> - Environmental Planner/Aerial Photographer, Ohio Dept. of Natural Resources, Columbus, Ohio.

SERVICE:

<u>2003-present</u> – Board of Directors, The Clearwater Conservancy, State College, Pennsylvania;
<u>2000-present</u> – Science and Education Advisory Committee, The ACEER Foundation, U.S.A. and Peru.

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